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Golder Associates Inc.

305 Fellowship Road, Suite 200
Mt. Laurel, NJ USA 08054
Tel: (609) 273-1110
Fax (609) 273-0778

Golder Associates

March 7, 1994

Project No.: 923-6112

United States Environmental Protection Agency (3HW24)
841 Chestnut Building
Philadelphia, PA 19107-4431

Attn: Mr. Frank Klanchar
Remedial Project Manager

RE: RUETGERS-NEASE SITE, STATE COLLEGE PA
ENVIRONMENTAL RISK ASSESSMENT

Gentlemen:

Further to our recent discussions we enclose a redline draft of the revised Environmental Risk Assessment. The following changes have been made to address Agency concerns, as agreed at our meeting on January 14, 1994:

1. A significant amount of additional data for Spring Creek has been summarized and presented in text, tables and figures:

   - Historical sediment chemistry data
   - Historical fish tissue residue data
   - Fish community data
   - Benthic macroinvertebrate community data

   Spatial and temporal trends in these data are discussed and the results are integrated into the risk characterization. Additional data from various studies such as the National Study for Chemical Residues in Fish (USEPA, 1992) have also been included to place the Spring Creek data in context.

2. Spring Creek biological community data (fish and benthic macroinvertebrates) have been compared to that of a "reference stream" (Fishing Creek) suggested by PaDER's Regional Aquatic Biologist.

3. References to the Sediment Toxicity Testing program have been revised to be consistent with the revised report transmitted to the Agency on January 26, 1994.

4. The Risk Characterization and Uncertainty sections have been revised and expanded to integrate all of the additional data and provide as comprehensive an assessment as possible.

Please call with any comments at your convenience.

Very truly yours,

GOLDER ASSOCIATES INC.

Dan Woltering

for/ P. Stephen Finn, C.Eng.
Associate

cc: Tamara Royer, Ruetgers-Nease Corporation
Dan Woltering, ENVIRON Corporation
Judi Durda, Weinberg Consulting Group

PSF:alk

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X. ENVIRONMENTAL RISK ASSESSMENT

A. Introduction

The objective of this environmental risk assessment is to screen for and characterize the potential risks to ecological resources from chemical substances associated with the Nease Chemical Superfund Site in State College, Pennsylvania. For the purposes of this assessment, the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) site is defined as the current Ruetgers-Nease Corporation property plus the off-site drainage areas into which the chemicals of interest may have migrated (the Site). Previously collected data on the condition of benthic macroinvertebrates and fish in Thornton Spring and Spring Creek, in addition to toxicity test results of the surface waters and sediments of Spring Creek were considered in this assessment, along with a screening-level analysis of exposure and risk to receptor species based on the results of the Remedial Investigation (RI) chemical analyses of surface water, sediments, soils and fish tissues. Field observations describing habitats and fish and wildlife sightings in the area were also factored into the assessment.

1. Scope and Approach

Based on the RI characterization analytical data, past operations at the facility, the environmental fate characteristics, and available ecotoxicological effects data for specific chemical substances, the assessment focuses primarily on mirex and kepone. However, all of the chemicals detected as part of the RI are identified and their environmental behavior is evaluated in the assessment. Further, all chemicals are considered indirectly as all are potential stressors in the bioassays conducted with Site water and sediments.

The assessment is consistent with the general guidance contained in the U.S. Environmental Protection Agency (USEPA) environmental risk assessment manual entitled "Risk Assessment Guidance for Superfund, Volume II - Environmental Evaluation Manual" (USEPA 1989b). Additional guidance with regard to ecotoxicological thresholds/criteria for aquatic and terrestrial wildlife species is taken from USEPA, U.S. Fish & Wildlife Service, and other published scientific reports.

Several different types of data are used in the overall assessment of potential risks associated with the Site. These are:

- chemical monitoring (both RI and historical);
- bioassay data;
- ecological resource information (i.e., habitat types, wildlife, and aquatic biota from both RI and historical surveys); and
- predicted exposure and toxicity values for a number of ecological receptors representative of the biota, exposure pathways, and specific chemical substances associated with the Site.

No one type of data is favored or used exclusively to assess risks. Rather, all of the available data is considered collectively to provide a comprehensive assessment of risk.

Analysis of the chemical monitoring data and an evaluation of exposure and potential risk were conducted for six distinctive zones or "Risk Management Units" (RMUs) within the Site area. The RMU approach is an overall conceptual framework used to guide the risk assessment process. The RMUs are distinct areas of the Site that are separated for the assessment because they are different habitats and/or are spatially separated and could therefore have different exposures to the chemicals of potential concern. The RMUs are identified on Figure 1 and consist of the following:

RMU1 - The approximately 15-acre grassy field located to the southwest of the developed (fenced in) portion of the Ruetgers-Nease property;

RMU2 - The drainage ditch from the point at which the Ruetgers-Nease ground water treatment facility effluent is discharged, downstream to the confluence of the ditch with Spring Creek (a distance of approximately 2,000 feet). For the risk characterization, RMU2 is further divided into the drainage ditch on Ruetgers-Nease property (RMU2A) and the drainage ditch beyond the property to the point where it enters Spring Creek (RMU2B).

RMU3 - Thornton Spring from the point at which it emerges from the ground to its confluence with Spring Creek (a distance of approximately 200 feet);

RMU4 - Spring Creek and its riparian zone in the vicinity of the Pike Street Bridge in Lemont (upstream of both the drainage ditch and Thornton Spring confluences);

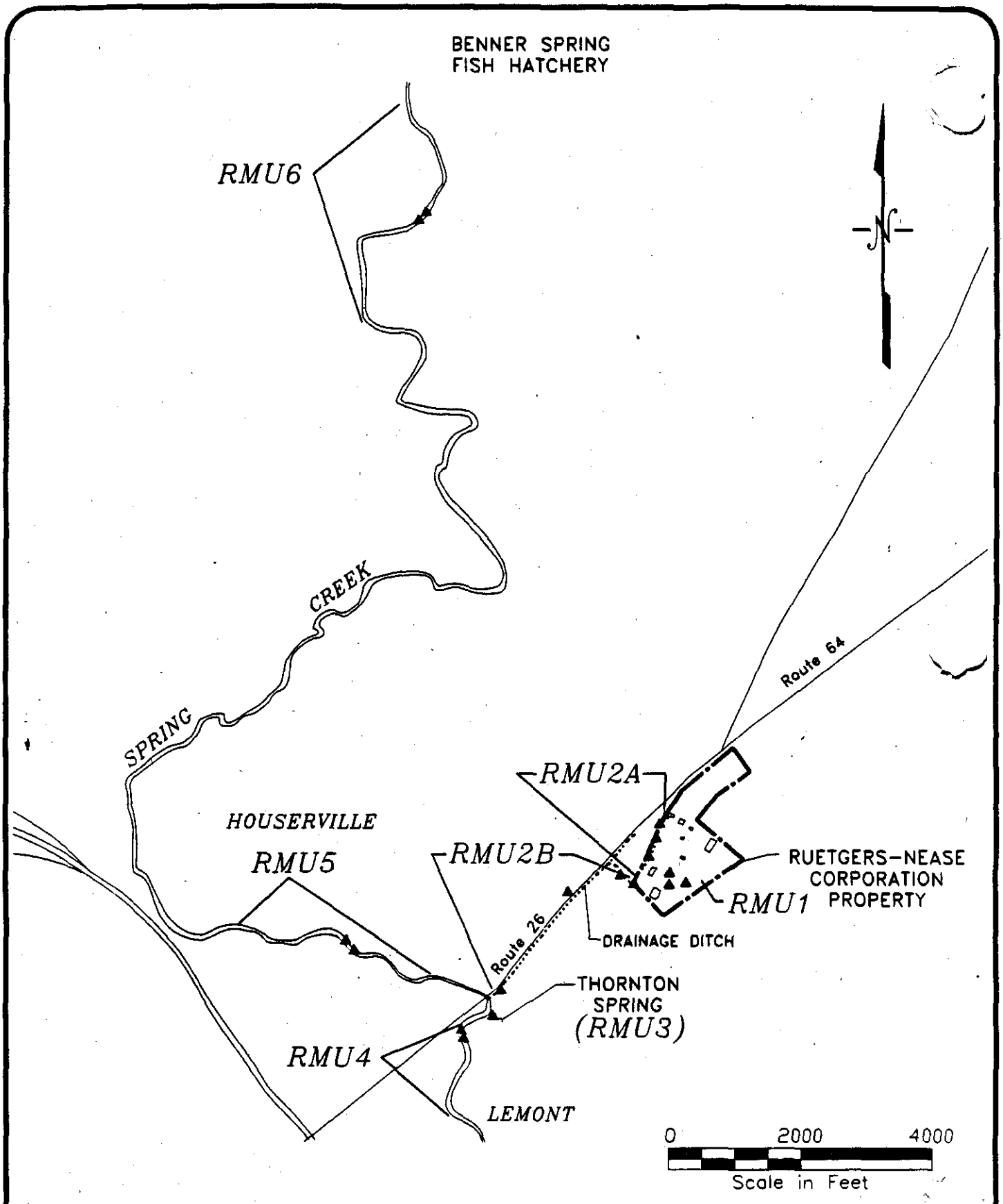
RMU5 - Spring Creek and its riparian zone in the vicinity of Houserville Park (downstream of both the drainage ditch and Thornton Spring confluences); and

RMU6 - Spring Creek and its riparian zone in the vicinity of the Pennsylvania Fish Commission Research Station and Hatchery at Benner Spring (further downstream from RMU5).

These RMUs were identified for sampling and analysis based upon their geographic locations relative to potential surface and subsurface sources of chemicals associated with the facility. All RMUs, except RMU4, are in the potential migration pathway for chemicals originating from the facility. RMU4 is upstream from the sources and therefore it serves as a "background area" for Spring Creek.

While the RMUs are considered as distinctive zones or areas, there are no distinct boundaries between the three Spring Creek RMUs other than their proximity to potential sources of chemical inputs. For example, chemical exposures are described by sampling conducted in one upstream (or background) area (i.e., RMU4), immediately downstream of where the drainage ditch and Thornton Spring enter Spring Creek (i.e., RMU5), and considerably further downstream (i.e., RMU6). Also, the estimates of chemical exposure to biota inhabiting an RMU-area are based on samples taken at one, or at most a few, points within the area. These sample points are considered to be representative of the surrounding area. Importantly, there are also historical data for fish and for benthic macroinvetebrates that are integrated into the risk assessment for those areas of Spring Creek that are between RMUs 4 and 5, between RMUs 5 and 6, and downstream from RMU6.

Ecological impacts are assessed separately for each RMU as a way to distinguish differences associated with habitats, exposure levels in soil, sediment and/or water, and distance from the source(s) of chemical input.



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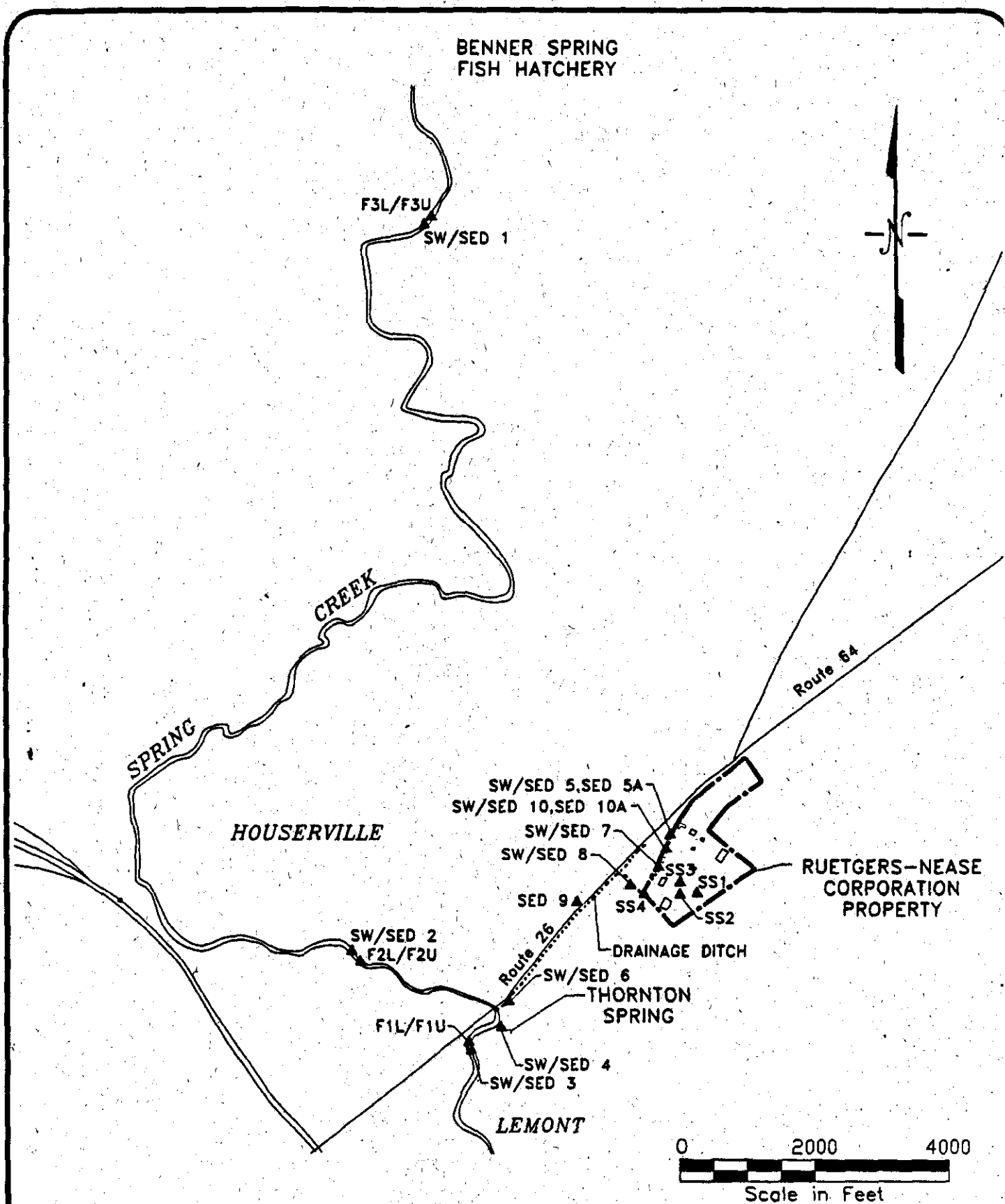
RISK MANAGEMENT UNIT (RMU) LOCATIONS
NEASE CHEMICAL SUPERFUND SITE
STATE COLLEGE, PENNSYLVANIA

Figure
1



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REMEDIAL INVESTIGATION SAMPLING LOCATIONS
NEASE CHEMICAL SUPERFUND SITE
STATE COLLEGE, PENNSYLVANIA

Figure 2

2. Organization of the Report

This environmental risk assessment is divided into seven sections as follows:

Section A - Introduction
Section B - Summary of Data to be Used in the Risk Assessment
Section C - Selection of Chemicals for Evaluation
Section D - Exposure Characterization
Section E - Toxicity Thresholds for Mirex and Kepone
Section F - Risk Characterization
Section G - Uncertainties in the Analysis

B. Summary of Data to be Used in the Risk Assessment

This section presents a summary of the analytical and bioassay data and the ecological resource information to be used in the risk assessment. Both RI and historical data are presented.

1. Chemistry Data: Remedial Investigation

SMC Environmental Services Group (SMC) conducted a site characterization on behalf of Ruetgers-Nease Corporation as part of the Remedial Investigation (RI). The design and implementation of these investigative studies have been approved by USEPA Region III. This work forms the basis for evaluating potential exposures to site-derived substances.

Summary statistics, including frequency of detection, minimum and maximum detected levels, and the range of reported detection limits for each target analyte on the Target Compound List (TCL) that was detected in various sampled media that are relevant to the ecological assessment, are presented in Appendix O (viz., surface soil (Table O-1), surface water (Table O-4), sediment (Table O-5), and fish (Table O-6)). The summary statistics are based upon sampling data provided by SMC. Table O-8 in Appendix O indicates those environmental media in which each of the target analytes were detected.

a) RMU-Specific Media Sampling and Chemical Analyses

The ecological exposure pathways are identified and the potential risks are assessed for each of the six RMUs. The sampling locations and sampling media are detailed below for each of the RMUs.

Surface soil was sampled at RMU1 and analyzed for kepone, mirex, and volatile organics (VOCs). Surface water and sediments were sampled from RMU2 - RMU6 and analyzed for kepone, mirex, and VOCs. Fish tissues were sampled from upper and lower trophic level fish taken at RMU4 - RMU6. These samples were analyzed for mirex and kepone. Figure 2 identifies the approximate locations of surface water, sediment, and soil samples for each RMU. Exact locations and descriptions of each sample along with the analytical data (see Figures 4-3, 4-5, and 4-6 *Phase I and Phase II Soil, Surface Water, and Sediment VOC, Kepone and Mirex Sampling Results*) are included in the RI. A brief summary follows.

1) **RMU1**

Surface soil samples were collected from the Ruetgers-Nease property outside of the fenced area, specifically in the area where the spray field was previously located (samples SS1, SS2, SS3). Five VOCs were detected at, or just above, their detection limit (i.e., 5-10 ppb). Both kepone and mirex were detected in surface soils (concentrations are shown in Table 62).

2) **RMU2A**

Surface water and/or sediments were collected from the drainage ditch which is further west and down-gradient from the west corner of the Ruetgers-Nease ground water treatment building (SW5, SW5-2/SW11 [a duplicate], SW7, SW10 and SED5, SED5-2/SED11 [a duplicate], SED7, SED10, and SS4). All of these sample locations are on Ruetgers-Nease property. Total VOCs were up to 4.5 ppm in surface water and up to 44.5 ppm in sediments. Mirex and kepone were detected in sediments (Table 62).

3) **RMU2B**

Surface water and/or sediment was also collected from the drainage ditch downgradient from the Ruetgers-Nease property, along Route 26, and just prior to its confluence with Spring Creek (SW8 and SED8, SED9, and SW6 and SED6, respectively). VOCs were at much lower concentrations in the off-site drainage ditch. Total VOCs were detected just at the 5 to 10 ppb detection limit in surface waters, and non-detect in sediments. Mirex and

kepone were also at lower concentrations in the off-site drainage ditch sediments (Table 62).

4) RMU3

Surface water and sediment were collected from Thornton Spring immediately upstream from the culvert at Pike Street and prior to its confluence with Spring Creek (SW4 and SED4). Total VOCs were at approximately 1.3 and 1.8 ppm in Thornton Spring water and sediments, respectively. Mirex and kepone were just above 0.5 ppm in sediments (Table 62).

5) RMU4

Surface water (SW3) and sediment (SED3) were collected from Spring Creek approximately 1,000 feet upstream from the confluence of Thornton Spring. VOCs, mirex and kepone were below detection in water and sediments.

Samples of fish tissue from a lower trophic level fish, the slimy sculpin (*Cottus cognatus*), were collected from Spring Creek at the Pike Street bridge in Lemont (F1L); again, upstream of the confluence of Thornton Spring. Fish tissues from a higher trophic level fish, the brown trout (*Salmo trutta*), were also collected from this upstream location (F1U). Fish were sampled for mirex and kepone; the results are presented in Table 62.

6) RMU5

Surface water (SW2) and sediment (SED2) were collected from Spring Creek in the area of Houserville Park (SW2). The presence of VOCs was not confirmed. Mirex and Kepone results are in Table 62. Slimy sculpin (F2L) and brown trout (F2U) fish tissue were taken from Spring Creek at Houserville Park. Fish sampling results are reported in Table 62.

7) RMU6

A surface water sample (SW1) and a sediment sample (SED1) were collected from Spring Creek at the Benner Spring Fish Hatchery. Total VOCs in the range of 100 ppb were reported in surface water. Mirex and kepone results are in Table 62. Slimy sculpin (F3L) and brown trout (F3U) fish

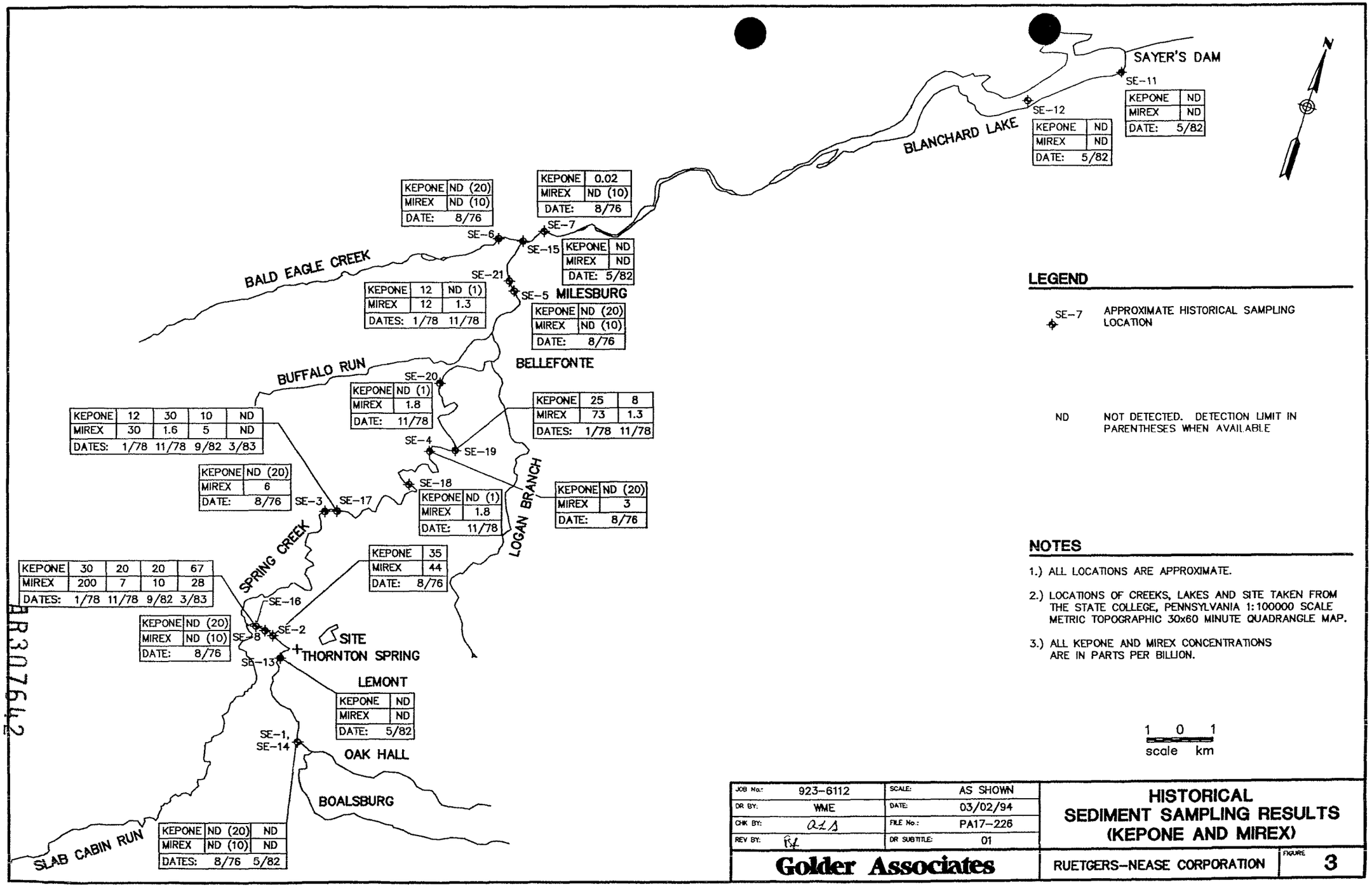
tissues were collected from this downstream location. Fish sampling results are reported in Table 62.

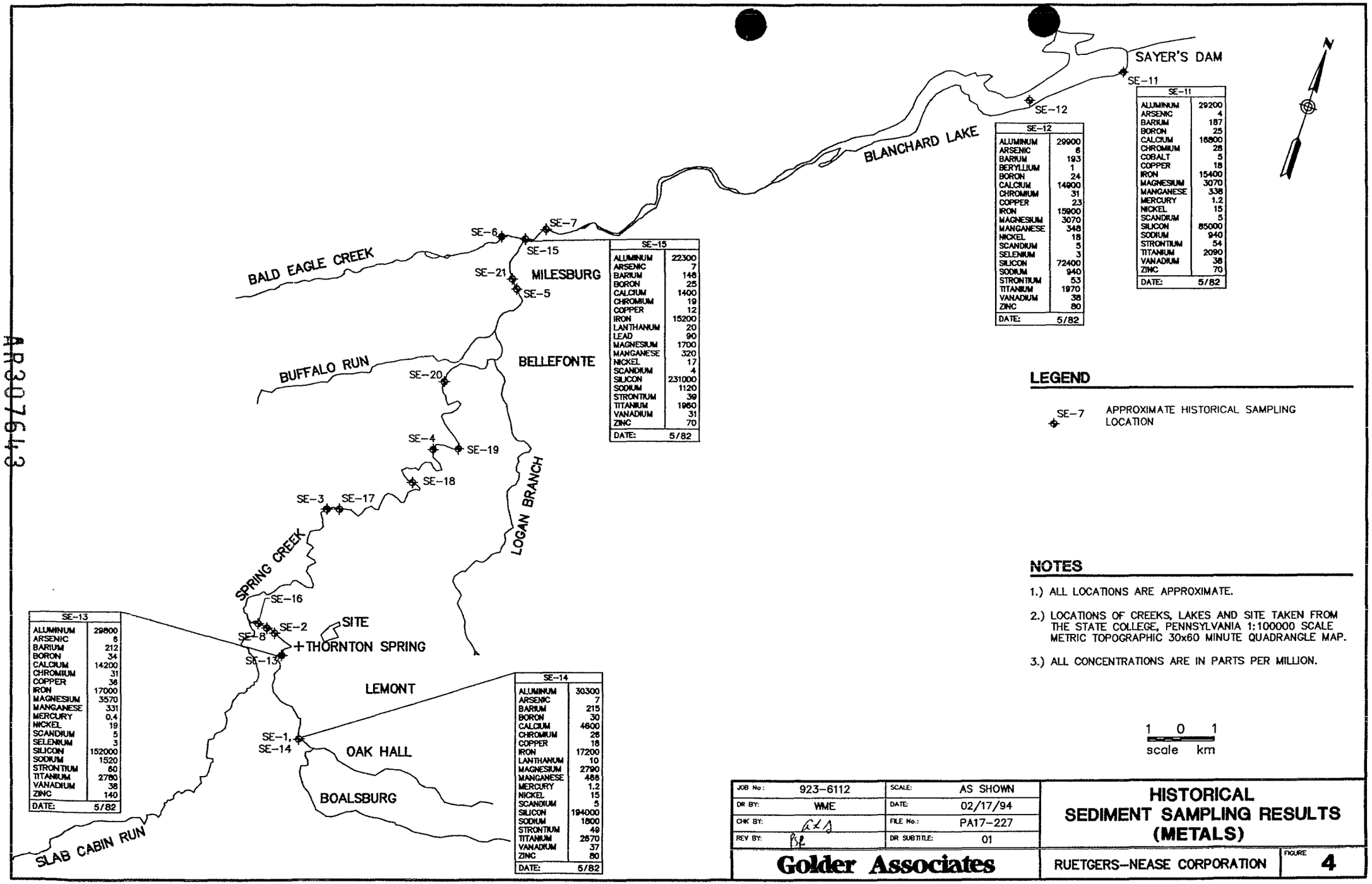
2. Chemistry Data: Historical

The historical data are useful for assessing spatial and temporal trends in the distribution and levels of chemicals in the Site area, but historical data were not used in the quantitative risk analysis. The RI chemistry data provides the best information on the current conditions and therefore are most appropriate to use in the quantitative risk assessment. The sampling and analytical methods on which the historical data are based were also different than those used in the RI, and often, the methods were not consistent with current quality control/quality assurance procedures. The available historical chemistry data for sediments and fish in Spring Creek are presented below.

a) Historical Sediment Data

The historical data available from previous sediment studies have been described in the RI revisions, pages 1-40.25 and 1-40.26 (October 1993). The mirex and kepone data collected during these studies is presented in Figure 3, which also shows the locations of the samples within the Spring Creek watershed relative to the Site. In one of the studies, conducted by USEPA in 1982, sediment samples were also analyzed for volatiles, semi-volatiles, and metals. USEPA reported that "no quantifiable concentration of any VOA, acid, or base/neutral fraction priority pollutant was found in any of the samples". The metals data provided by the study is presented in Figure 4 which also shows the locations of the samples within the Spring Creek watershed relative to the Site. USEPA reported that there was "no evidence of elevated levels of heavy elements in either fish or sediments" in the 1982 study.


SAYER'S DAM
SE-11
KEPONE ND
MIREX ND
DATE: 5/82
SE-12
KEPONE ND
MIREX ND
DATE: 5/82
BLANCHARD LAKE
N
KEPONE ND (20)
MIREX ND (10)
DATE: 8/76
KEPONE 0.02
MIREX ND (10)
DATE: 8/76
SE-6
SE-7
SE-15
KEPONE ND
MIREX ND
DATE: 5/82
BALD EAGLE CREEK
SE-21
SE-5
MILESBURG
KEPONE 12 ND (1)
MIREX 12 1.3
DATES: 1/78 11/78
KEPONE ND (20)
MIREX ND (10)
DATE: 8/76
BELLEFONTE
BUFFALO RUN
SE-20
KEPONE ND (1)
MIREX 1.8
DATE: 11/78
KEPONE 25 8
MIREX 73 1.3
DATES: 1/78 11/78
KEPONE 12 30 10 ND
MIREX 30 1.6 5 ND
DATES: 1/78 11/78 9/82 3/83
SE-4
SE-19
SE-18
KEPONE ND (1)
MIREX 1.8
DATE: 11/78
KEPONE ND (20)
MIREX 3
DATE: 8/76
LOGAN BRANCH
KEPONE ND (20)
MIREX 6
DATE: 8/76
SE-3
SE-17
SPRING CREEK
KEPONE 35
MIREX 44
DATE: 8/76
KEPONE 30 20 20 67
MIREX 200 7 10 28
DATES: 1/78 11/78 9/82 3/83
SE-16
KEPONE ND (20)
MIREX ND (10)
DATE: 8/76
SE-8
SE-2
SITE
THORNTON SPRING
SE-13
LEMONT
KEPONE ND
MIREX ND
DATE: 5/82
SE-1,
SE-14
OAK HALL
BOALSBURG
SLAB CABIN RUN
KEPONE ND (20) ND
MIREX ND (10) ND
DATES: 8/76 5/82
LEGEND
SE-7 APPROXIMATE HISTORICAL SAMPLING LOCATION
ND NOT DETECTED. DETECTION LIMIT IN PARENTHESES WHEN AVAILABLE
NOTES
1.) ALL LOCATIONS ARE APPROXIMATE.
2.) LOCATIONS OF CREEKS, LAKES AND SITE TAKEN FROM THE STATE COLLEGE, PENNSYLVANIA 1:100000 SCALE METRIC TOPOGRAPHIC 30x60 MINUTE QUADRANGLE MAP.
3.) ALL KEPONE AND MIREX CONCENTRATIONS ARE IN PARTS PER BILLION.
1 0 1
scale km
JOB No.: 923-6112
SCALE: AS SHOWN
DR BY: WME
DATE: 03/02/94
CHK BY: AJS
FILE No.: PA17-226
REV BY: RA
DR SUBTITLE: 01
Golder Associates
HISTORICAL SEDIMENT SAMPLING RESULTS (KEPONE AND MIREX)
RUETGERS-NEASE CORPORATION
FIGURE 3



LEGEND

SE-7 — APPROXIMATE HISTORICAL SAMPLING LOCATION

NOTES

1.) ALL LOCATIONS ARE APPROXIMATE.

2.) LOCATIONS OF CREEKS, LAKES AND SITE TAKEN FROM THE STATE COLLEGE, PENNSYLVANIA 1:100000 SCALE METRIC TOPOGRAPHIC 30x60 MINUTE QUADRANGLE MAP.

3.) ALL CONCENTRATIONS ARE IN PARTS PER MILLION.

1 0 1
scale km

JOB No:	923-6112	SCALE:	AS SHOWN
DR BY:	WME	DATE:	02/17/94
CHK BY:		FILE No.:	PA17-227
REV BY:		DR SUBTITLE:	01

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HISTORICAL SEDIMENT SAMPLING RESULTS (METALS)

RUETGERS-NEASE CORPORATION

FIGURE 4

Certain trends are apparent from a review of the historical sediment concentrations for mirex and kepone. Figures 5 through 7 show mirex and kepone concentrations in sediment as a function of distance from the Site for each sampling event in which multiple samples were analyzed (i.e., August 1976, January 1978, and November 1978, respectively). In each case there is a trend of decreasing concentration with increasing distance from the Site. Temporal trends in the data are less obvious, although there is an indication that sediment concentrations of mirex and kepone may be declining over time. The metals data presented in Figure 4 show no apparent spatial trends and the levels are generally close to the mean, and all well within the ranges, for background concentrations in the eastern United States as reported by Shacklette and Boerngen (1984).

b) Historical Fish Data

Fish tissue samples have been collected from Spring Creek for chemical analysis since 1976. The analytical efforts have focused on mirex and kepone concentrations in native fish, although other chemicals were included in a few of the studies. The historical data available for each of the fish tissue sampling efforts in Spring Creek are presented in the 1993 RI Revisions (pages 1-40.26 through 1-40.43 plus Table 1-9). A summary of these historical fish tissue sampling efforts is shown in Table 54.

The fish sampling efforts have included a number of stations, but those downstream of the Site that have been sampled in three or more years are of the most use in evaluating historical trends relevant to this assessment. The stations include PADER collection locations (see Figure 10): Station 1 (map locator FT-10) in Houserville about 0.5 mile downstream from the Route 26 Bridge; Station 2 (map locator FT-11) adjacent to the Benner Spring Hatchery; Station 2.5 (map locator FT-12) adjacent to the Upper Spring Creek Hatchery; Station 3 (map locator FT-13) just downstream from Fisherman's Paradise; Station 3.5 (map locator FT-14) located just upstream of the Route 550 Bridge; and Station 4 (map locator FT-15) just downstream from McCoy Dam. The available historical data for each of these sampling stations are listed in Tables 55 through 60, respectively.

Table 54
Summary of Historical Fish Tissue Sampling Efforts in Spring Creek

STUDY	SAMPLE DATE(S)	NUMBER OF SAMPLE LOCATIONS	FISH SPECIES
PaDER/Pennsylvania Fish Commission (PFC). 1978	August 30-31, 1976	9 (a)	Brown trout White sucker
PaDER. 1979	November 20-21, 1978	6	Brown trout White sucker
USEPA-National Enforcement Investigations Center (NEIC). 1979	January 1978 and November 1978	2	Brown trout
Center for Bio-Organic Studies (CBOS). 1979	July 9-10, 1979	8 (b)	Brown trout White sucker
Center for Bio-Organic Studies (CBOS). 1980	July 9-10, 1979	1	Brown trout White sucker
PaDER. 1980	July 1979	8	Brown trout
PaDER. 1982	September 30, 1982	2	Brown trout Catfish Slimy sculpin White sucker
PaDER. 1985, 1986, 1987.	May 1976 - October 1986	8	Brown trout Catfish Slimy sculpin White sucker

(a) One of the nine sampling locations was a control station located at the Spring Creek Hatchery.
(b) Data was reported for only three of the eight sampling stations in the study.

Table 55
Summary of Historical Fish Tissue Data from the PaDER 1985, 1986, 1987 Study

Sample Location (a)	Distance (miles) (b)	Sample Year	Species (c)	Trophic Level (d)	Concentration (ppb) (e) Fillets with Skin	
					Kepone	Mirex
10	0.5	1978	BT	UT	116	2970
10 (f)	0.5	1978	WS	LT	69 (g)	245 (g)
11	5	1978	BT	UT	50	48
11 (h)	5	1978	WS	LT	12 (g)	112 (g)
12	7.4	1978	BT	UT	(i)	(i)
12	7.4	1978	WS	LT	ND (g)	66 (g)
13	9.9	1978	BT	UT	36	168
13 (j)	9.9	1978	WS	LT	32 (g)	104 (g)
14	11.9	1978	BT	UT	(i)	(i)
14	11.9	1978	WS	LT	ND (g)	52 (g)
15	15	1978	BT	UT	ND	44
15 (k)	15	1978	WS	LT	9	56

ND = Not Detected

(a) Sample locations are depicted in Figure 10.

(b) All distances are downstream from Route 26 Bridge.

(c) BT = Brown trout, WS = White sucker

(d) UT = Upper trophic level, LT = Lower trophic level

(e) Concentration represents whole body or fillets with skin, unless otherwise noted.

(g) Concentration represents fillets with no skin.

(f) Kepone value presented is from the 1/78 sample results. Mirex value presented is the average of 1/78 (250 ppb) and 11/78 (240 ppb) sample results.

(h) Kepone value presented is from the 1/78 sample results. Mirex value presented is the average of 1/78 (160 ppb) and 11/78 (64 ppb) sample results.

(i) No data available.

(j) Kepone value presented is from the 1/78 sample results. Mirex value presented is the average of 1/78 (160 ppb) and 11/78 (48 ppb) sample results.

(k) Kepone value presented is from the 1/78 sample results. Mirex value presented is the average of 1/78 (95 ppb) and 11/78 (16 ppb) sample results.

Table 56
Summary of Historical Fish Tissue Data for the Houserville Area of Spring Creek

Sample Location (a)	Reference	Sample Year	Species (b)	Trophic Level (c)	Concentration (ppb) (d) Fillets with Skin	
					Kepone	Mirex
10 (e)	PaDER. 1985, 1986, 1987.	1976	BT	UT	180	1000
10	PaDER. 1985, 1986, 1987.	1976	WS	LT	230 (f)	750 (f)
10 (g)	PaDER. 1979, USEPA. 1979, PaDER. 1985, 1986, 1987.	1978	BT	UT	116	1759
10 (h)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	WS	LT	69 (f)	243 (f)
10 (i)	CBOS. 1979, CBOS. 1980, PaDER. 1980	1979	BT	UT	130	908
10	CBOS. 1979, CBOS. 1980	1979	WS	LT	ND (f)	100 (f)
10	PaDER. 1985, 1986, 1987.	1981	BT	UT	270	630
10 (j)	PaDER. 1982, PaDER. 1985, 1986, 1987.	1982	BT	UT	30	250
10 (k)	PaDER. 1982, PaDER. 1985, 1986, 1987.	1982	CF, SS	LT	450	153
10 (l)	PaDER. 1985, 1986, 1987.	1983	BT	UT	ND	260
10 (m)	PaDER. 1985, 1986, 1987.	1983	CF, SS	LT	198	145
10 (n)	PaDER. 1985, 1986, 1987.	1984	BT	UT	ND	89
10	PaDER. 1985, 1986, 1987.	1984	SS	LT	200	48
10	PaDER. 1985, 1986, 1987.	1985	BT	UT	ND	110
10	PaDER. 1985, 1986, 1987.	1986	BT	UT	25	160

ND = Not Detected

(a) Sample locations are depicted in Figure 10.

(b) BT = Brown trout, CF = Catfish, SS = Slimy sculpin, WS = White sucker

(c) UT = Upper trophic level, LT = Lower trophic level

(d) Concentration represents whole body or fillets with skin, unless otherwise noted.

(e) Kepone value presented is the average of 5/76 (170 ppb) and 8/76 (190 ppb) sample results.

(f) Concentration represents fillets with no skin.

(g) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (585 ppb), USEPA 1979 (2800, 680 ppb) and PaDER 1985, 1986, 1987 (2970 ppb) studies.

(h) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (240 ppb) and PaDER 1985, 1986, 1987 (250, 240 ppb) studies.

(i) Kepone value presented is from the PaDER 1980 study. Mirex value presented is the average of CBOS 1979, CBOS 1980 (615 ppb) and PaDER 1980 (1200 ppb) studies.

(j) Kepone and mirex values presented in the PaDER 1982 study are equivalent to the kepone and mirex values presented in the PaDER 1985, 1986, 1987 study.

(k) Kepone value presented is the average of PaDER 1982 (660, 30 ppb) and PaDER 1985, 1986, 1987 (660 ppb) studies. Mirex value presented is the average of PaDER 1982 (210, 40 ppb) and PaDER 1985, 1986, 1987 (210 ppb) studies.

(l) Mirex value presented is the average of 3/83 (240 ppb) and 10/83 (280 ppb) sample results.

(m) Kepone value presented is the average of 3/83 (66 ppb) and 10/83 (480, 48 ppb) sample results. Mirex value presented is the average of 3/83 (45 ppb) and 10/83 (290, 100 ppb) sample results.

(n) Mirex value presented is the average of 3/84 (38 ppb) and 11/84 (140 ppb) sample results.

Table 57
Summary of Historical Fish Tissue Data for the Benner Spring Area of Spring Creek

Sample Location (a)	Reference	Sample Year	Species (b)	Trophic Level (c)	Concentration (ppb) (d) Fillets with Skin	
					Kepone	Mirex
11	PaDER. 1985, 1986, 1987.	1976	BT	UT	87	320
11	PaDER. 1985, 1986, 1987.	1976	WS	LT	31 (e)	61 (e)
11 (f)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	BT	UT	50	35
11 (g)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	WS	LT	12 (e)	96 (e)
11 (h)	CBOS. 1979, CBOS. 1980, PaDER. 1980	1979	BT	UT	ND	170
11	CBOS. 1979, CBOS. 1980	1979	WS	LT	(i)	65 (e)
11	PaDER. 1985, 1986, 1987.	1981	BT	UT	70	140
11 (j)	PaDER. 1982, PaDER. 1985, 1986, 1987.	1982	BT	UT	ND	120
11 (k)	PaDER. 1985, 1986, 1987.	1983	BT	UT	ND	200
11 (l)	PaDER. 1985, 1986, 1987.	1984	BT	UT	ND	60
11	PaDER. 1985, 1986, 1987.	1985	BT	UT	ND	68
11	PaDER. 1985, 1986, 1987.	1986	BT	UT	13	80

ND = Not Detected

(a) Sample locations are depicted in Figure 10.

(b) BT = Brown trout, WS = White sucker

(c) UT = Upper trophic level, LT = Lower trophic level

(d) Concentration represents whole body or fillets with skin, unless otherwise noted.

(e) Concentration represents fillets with no skin.

(f) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (22 ppb) and PaDER 1985, 1986, 1987 (48 ppb) studies.

(g) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (64 ppb) and PaDER 1985, 1986, 1987 (160, 64 ppb) studies.

(h) Kepone value presented is from the PaDER 1980 study. Mirex value presented is the average of CBOS 1979, CBOS 1980 (130 ppb) and PaDER 1980 (210 ppb) studies.

(i) No data available.

(j) Kepone and mirex values presented in the PaDER 1982 study are equivalent to the kepone and mirex values presented in the PaDER 1985, 1986, 1987 study.

(k) Mirex value presented is the average of 3/83 (240 ppb) and 10/83 (160 ppb) sample results.

(l) Mirex value presented is the average of 3/84 (30 ppb) and 11/84 (90 ppb) sample results.

Table 58
Summary of the Historical Fish Tissue Data for Locations FT-12 and FT-14 of Spring Creek

Sample Location (a)	Reference	Sample Year	Species (b)	Trophic Level (c)	Concentration (ppb) (d) Fillets with Skin	
					Kepone	Mirex
12	PaDER. 1979	1978	BT	UT	ND	285
12 (e)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	WS	LT	ND (f)	66 (f)
12	PaDER. 1980	1979	BT	UT	ND	350
12	PaDER. 1985, 1986, 1987.	1981	BT	UT	51	83
14	PaDER. 1979	1978	BT	UT	ND	130
14 (g)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	WS	LT	ND (f)	52 (f)
14	PaDER. 1980	1979	BT	UT	ND	270
14	PaDER. 1985, 1986, 1987.	1981	BT	UT	40	220

ND = Not Detected

(a) Sample locations are depicted in Figure 10.

(b) BT = Brown trout, WS = White sucker

(c) UT = Upper trophic level, LT = Lower trophic level

(d) Concentration represents whole body or fillets with skin, unless otherwise noted.

(e) Kepone and mirex values presented in the PaDER 1979 study are equivalent to the kepone and mirex values presented in the PaDER 1985, 1986, 1987 study.

(f) Concentration represents fillets with no skin.

(g) Kepone and mirex values presented in the PaDER 1979 study are equivalent to the kepone and mirex values presented in the PaDER 1985, 1986, 1987 study.

Table 59
Summary of Historical Fish Tissue Data for the Fisherman's Paradise Area of Spring Creek

Sample Location (a)	Reference	Sample Year	Species (b)	Trophic Level (c)	Concentration (ppb) (d) Fillets with Skin	
					Kepone	Mirex
13	PaDER. 1985, 1986, 1987.	1976	BT	UT	60	165
13	PaDER. 1985, 1986, 1987.	1976	WS	LT	40 (e)	46 (e)
13 (f)	PaDER. 1979, USEPA. 1979, PaDER. 1985, 1986, 1987.	1978	BT	UT	36	221
13 (g)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	WS	LT	32 (e)	85 (e)
13 (h)	CBOS. 1979, CBOS. 1980, PaDER. 1980	1979	BT	UT	ND	153
13	CBOS. 1979, CBOS. 1980	1979	WT	LT	(i)	155 (e)
13	PaDER. 1985, 1986, 1987.	1981	BT	UT	60	130

ND = Not Detected

(a) Sample locations are depicted in Figure 10.

(b) BT = Brown trout, WS = White sucker

(c) UT = Upper trophic level, LT = Lower trophic level

(d) Concentration represents whole body or fillets with skin, unless otherwise noted.

(e) Concentration represents fillets with no skin.

(f) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (175 ppb), USEPA 1979 (200, 340 ppb) and PaDER 1985, 1986, 1987 (168 ppb) studies.

(g) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (48 ppb) and PaDER 1985, 1986, 1987 (160, 48 ppb) studies.

(h) Kepone value presented is from the PaDER 1980 study. Mirex value presented is the average of CBOS 1979, CBOS 1980 (145 ppb) and PaDER 1980 (160 ppb) studies.

(i) No data available.

Table 60
Summary of Historical Fish Tissue Data for the McCoy Dam Area of Spring Creek

Sample Location (a)	Reference	Sample Year	Species (b)	Trophic Level (c)	Concentration (ppb) (d) Fillets with Skin	
					Kepone	Mirex
15	PaDER. 1985, 1986, 1987.	1976	BT	UT	37	88
15	PaDER. 1985, 1986, 1987.	1976	WS	LT	ND (e)	88 (e)
15 (f)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	BT	UT	ND	33
15 (g)	PaDER. 1979, PaDER. 1985, 1986, 1987.	1978	WS	LT	9 (e)	42 (e)
15	PaDER. 1980	1979	BT	UT	ND	150
15	PaDER. 1985, 1986, 1987.	1981	BT	UT	27	85
15	PaDER. 1985, 1986, 1987.	1984	BT	UT	ND	ND
15	PaDER. 1985, 1986, 1987.	1985	BT	UT	ND	35
15	PaDER. 1985, 1986, 1987.	1986	BT	UT	ND	ND

ND = Not Detected

(a) Sample locations are depicted in Figure 10.

(b) BT = Brown trout, WS = White sucker

(c) UT = Upper trophic level, LT = Lower trophic level

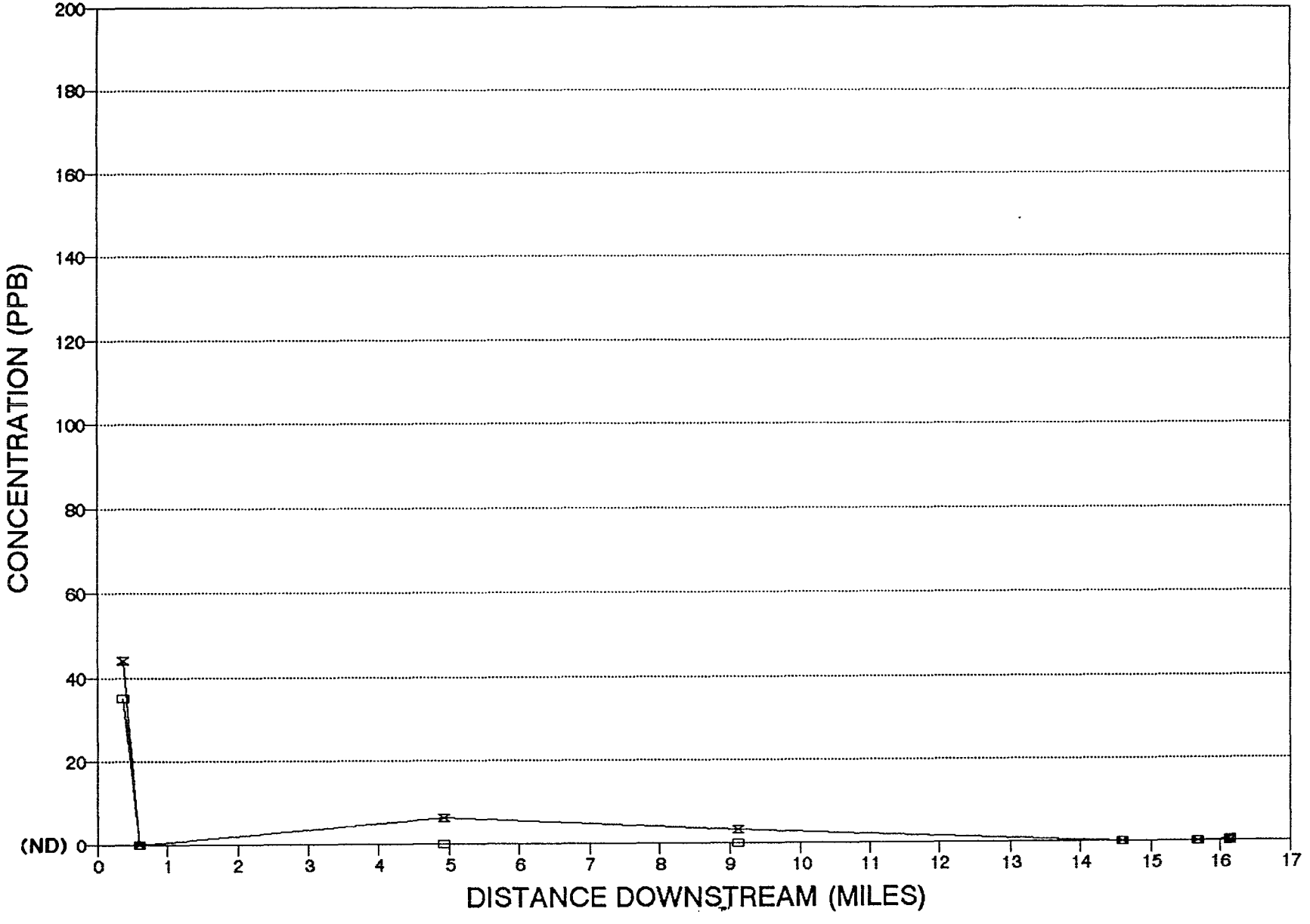
(d) Concentration represents whole body or fillets with skin, unless otherwise noted.

(e) Concentration represents fillets with no skin.

(f) Mirex value presented is the average of PaDER 1979 (22 ppb) and PaDER 1985, 1986, 1987 (44 ppb) studies.

(g) Kepone value presented is from the PaDER 1985, 1986, 1987 study. Mirex value presented is the average of PaDER 1979 (16 ppb) and PaDER 1985, 1986, 1987 (95, 16 ppb) studies.

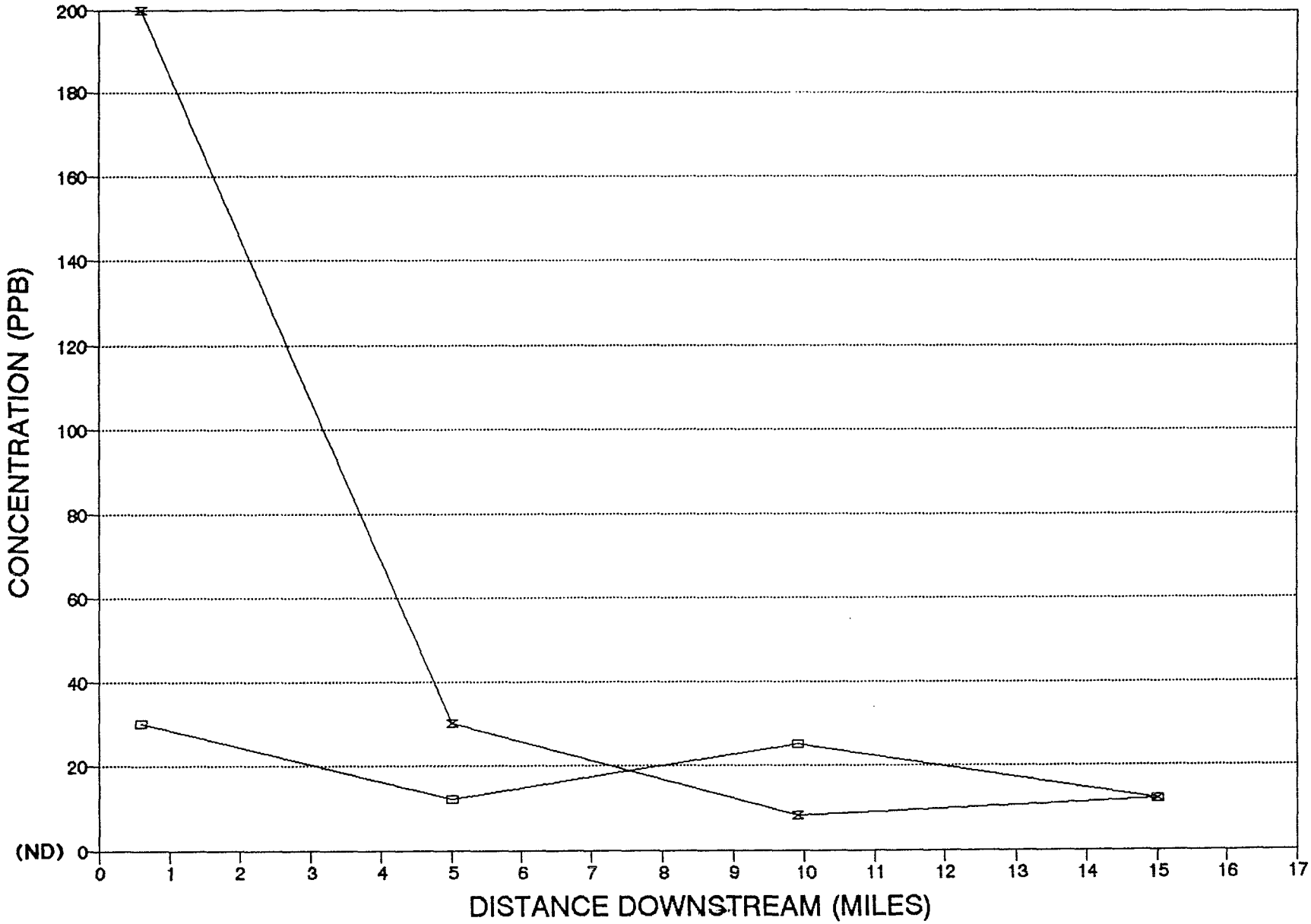
MIREX & KEPONE CONCENTRATIONS IN SEDIMENT AS A FUNCTION OF DISTANCE FROM THE SITE (AUGUST 1976)



ND = NOT DETECTED

FIGURE 5

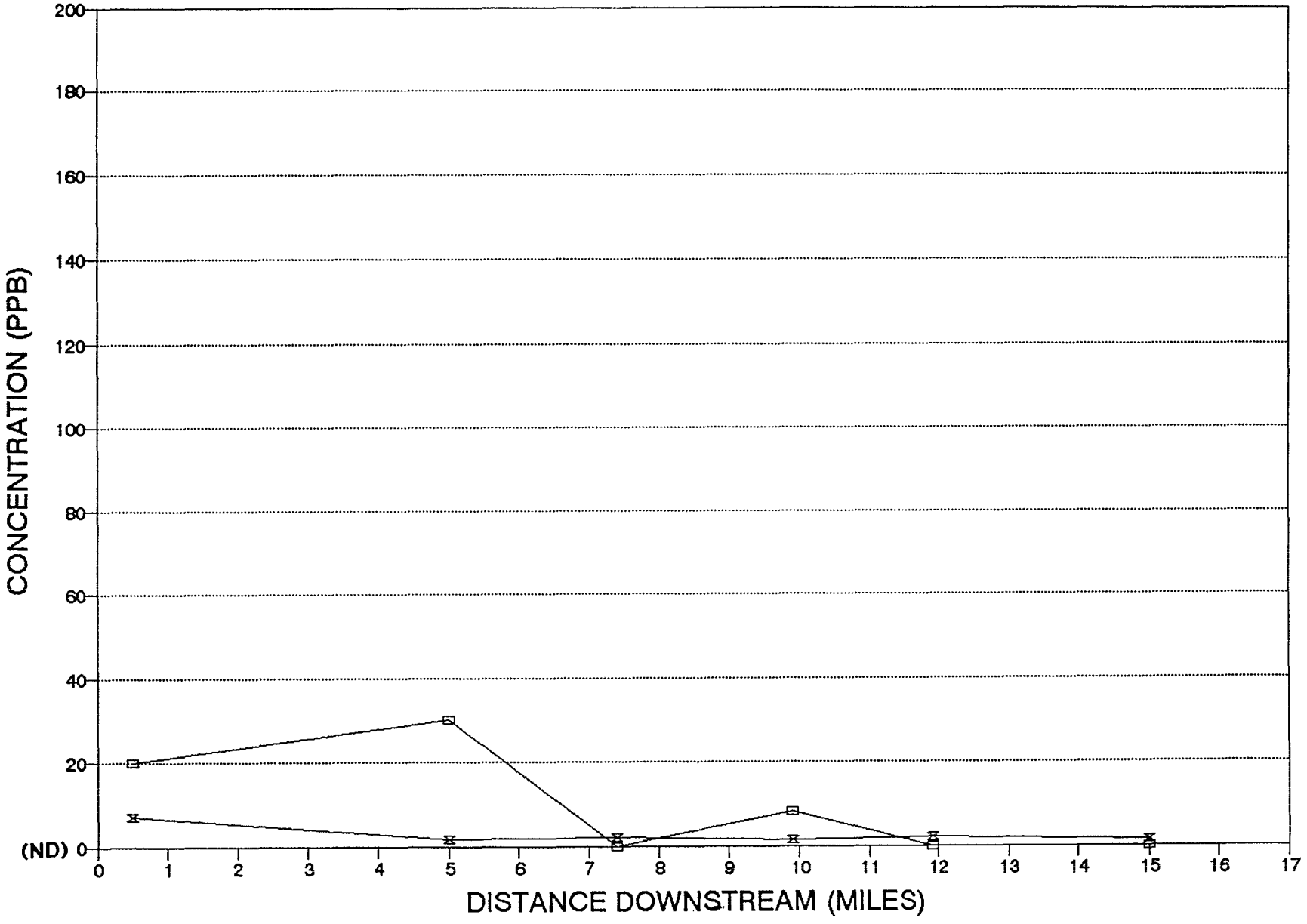
MIREX & KEPONE CONCENTRATIONS IN SEDIMENT AS A FUNCTION OF DISTANCE FROM THE SITE (JANUARY 1978)



ND = NOT DETECTED

FIGURE 6

MIREX & KEPONE CONCENTRATIONS IN SEDIMENT AS A FUNCTION OF DISTANCE FROM THE SITE (NOVEMBER 1978)



ND = NOT DETECTED

FIGURE 7

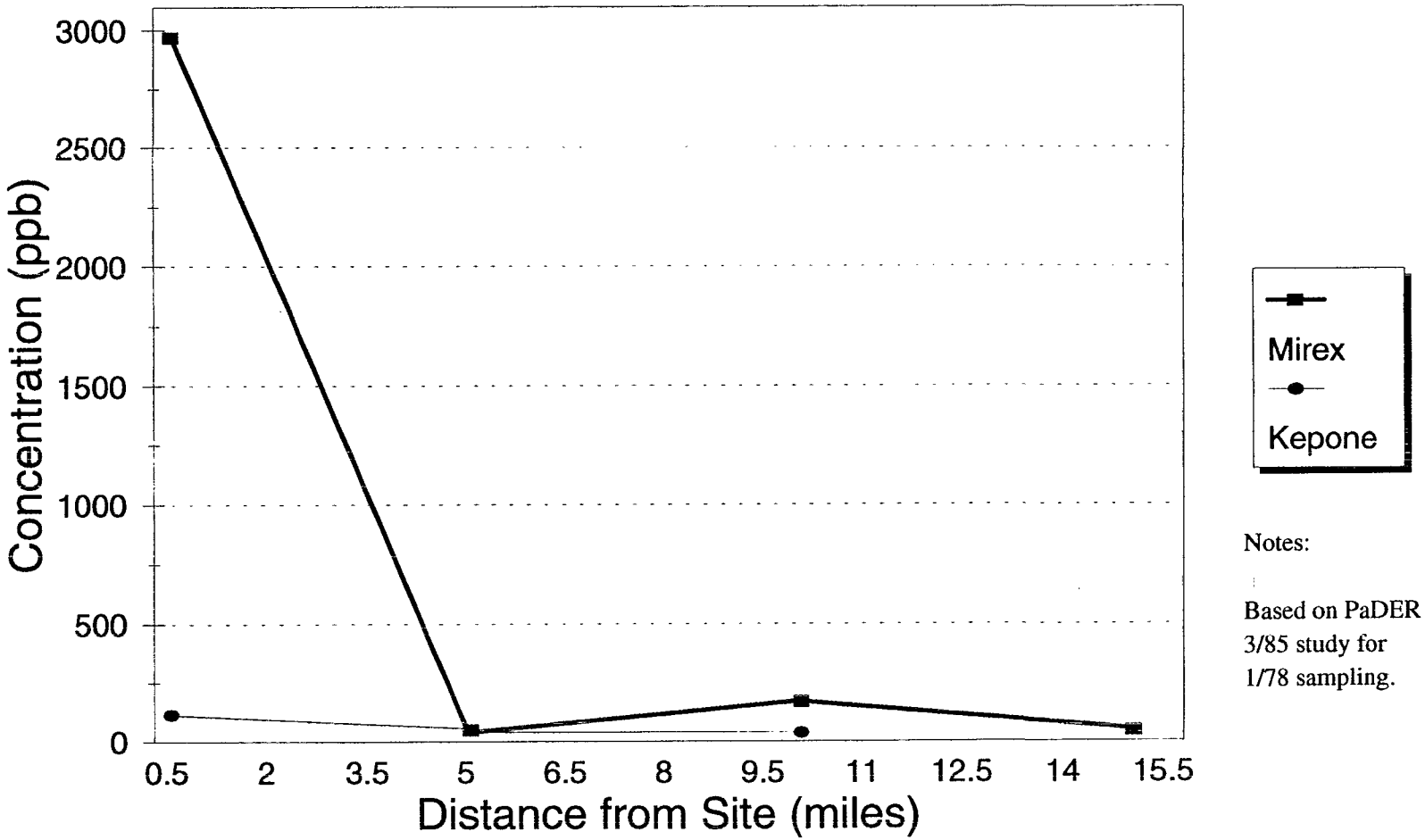
Several trends are apparent when reviewing the historical data for fish tissue concentrations for mirex and kepone. First, mirex and kepone concentrations in fish tissue decrease with increasing distances downstream from the Site. Second, mirex and kepone concentrations in fish tissue have decreased over time since they were first reported in the mid-1970s. Third, mirex and kepone concentrations in upper trophic level fish are often greater than the levels in lower trophic level fish. Finally, mirex concentrations generally exceed kepone concentrations.

Figures 8 and 9 (and Table 55) illustrate trends in mirex and kepone concentrations as a function of downstream distance from the Site. Brown trout (upper trophic level) tissue concentrations are from skin-on fillets or whole organisms, and white sucker (lower trophic level) tissue concentrations are from fillets with the skin off (because skin-on or whole fish data were not available). For both species, the tissue concentrations of mirex and kepone decrease with increasing distance downstream. Mirex concentrations show the greatest decrease with distance. Mirex concentrations in upper trophic level fish are almost 70 times lower in the most downstream sample compared to the levels reported for the fish collected nearest the Site. Those in lower trophic fish are approximately 4 times lower. Kepone concentrations in upper and lower trophic fish are more than 7 times lower in the most distant downstream sample.

Figure 10 (and Tables 56 - 60) shows the patterns of mirex and kepone concentrations in fish tissues collected over time, from 1976 to 1986, at stations where there were at least three years of data. The data for fish of both trophic levels generally show decreasing concentrations over time.

In addition to the temporal and spatial pattern of mirex and kepone in Spring Creek fish, there is historical information on the levels of pesticides, including mirex, in nationwide studies of fish tissue. This information can provide a point of reference for the Spring Creek data; however, the assessment of risk for this Site is appropriately based on those data collected on local fish.

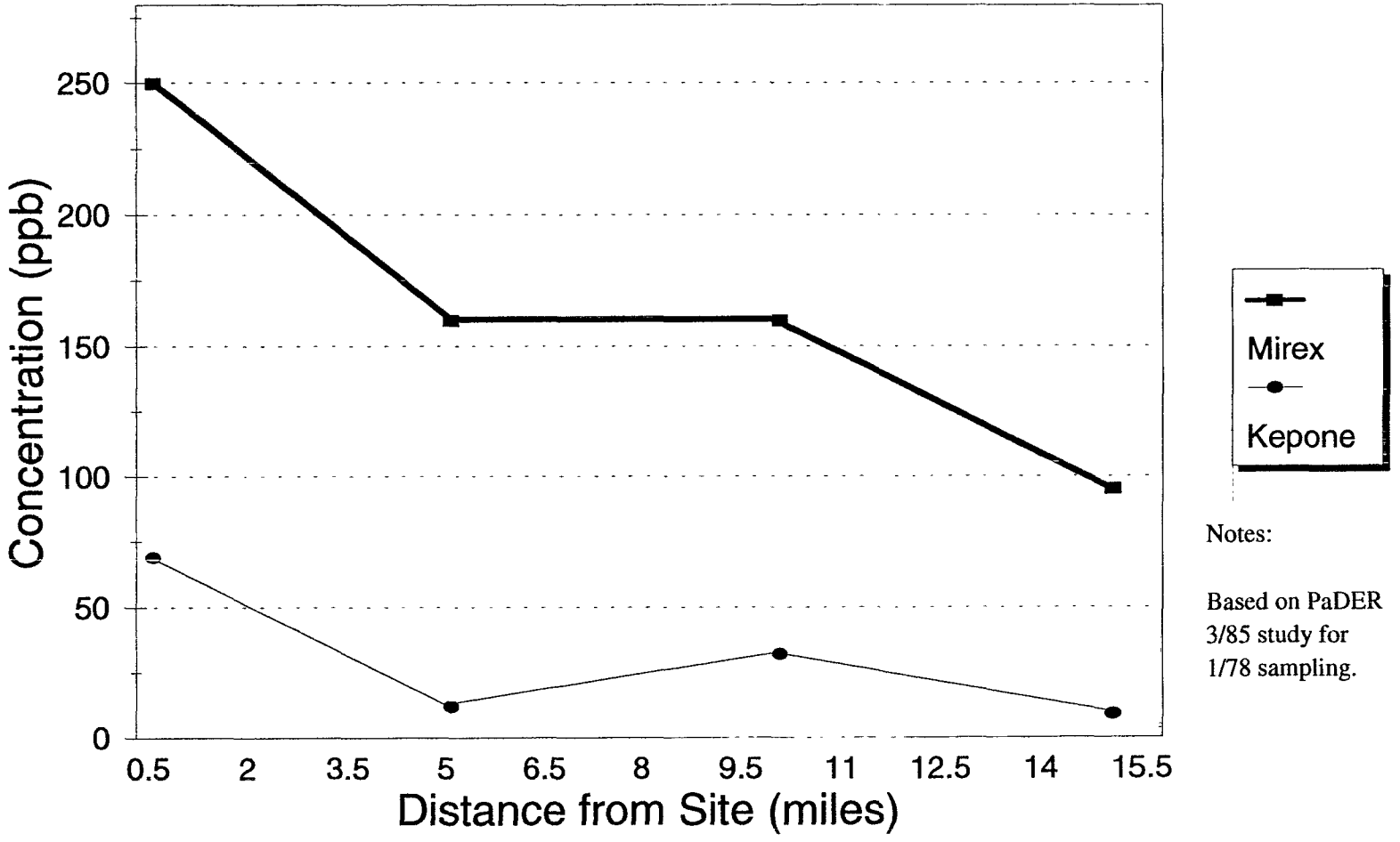
Figure 8
Fish Tissue Concentration as a Function of Distance from the Site (1978)
Upper Trophic Level



Notes:

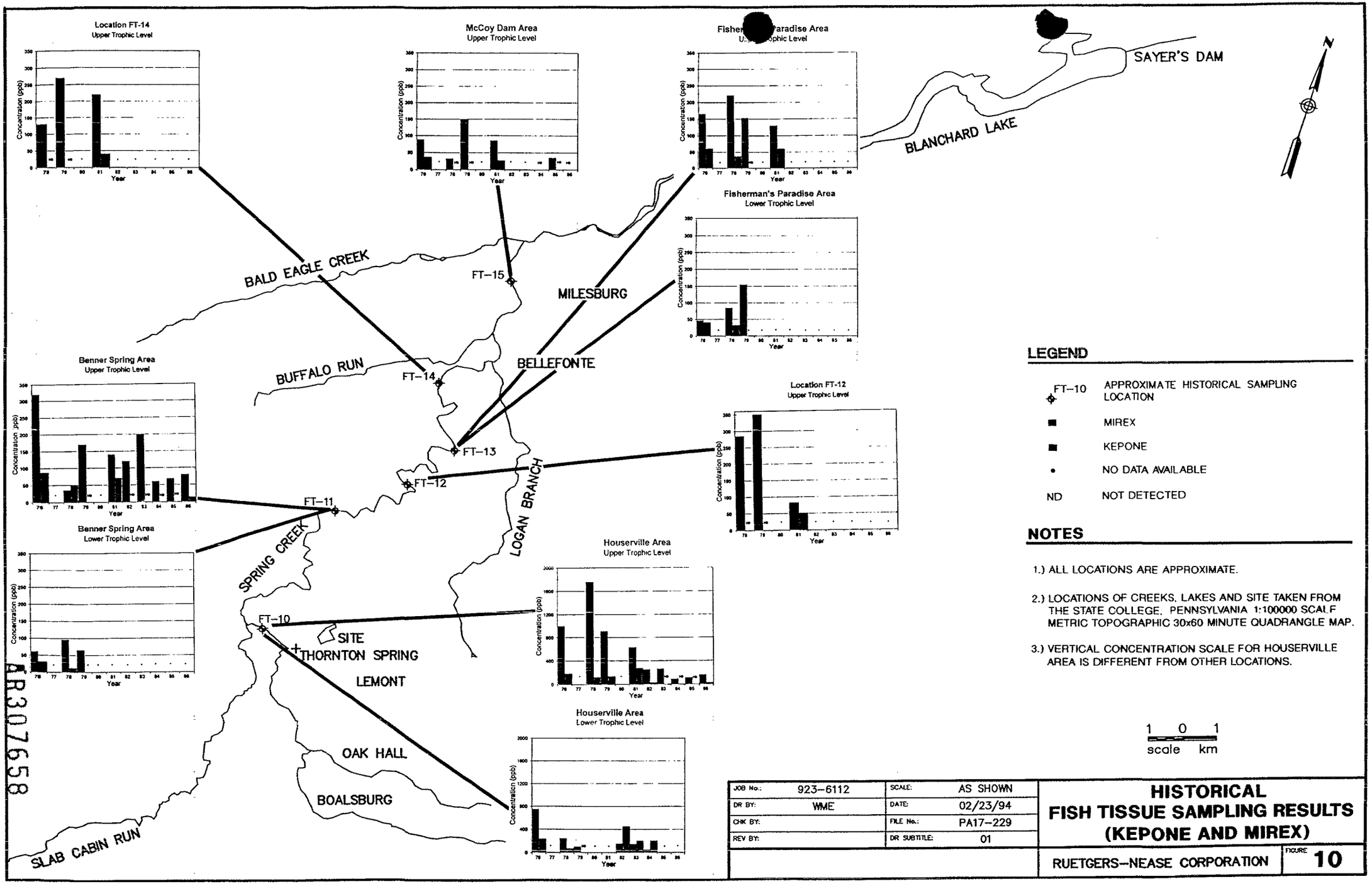
Based on PaDER 3/85 study for 1/78 sampling.

Figure 9
Fish Tissue Concentration as a Function of Distance from the Site (1978) Lower Trophic Level



Notes:

Based on PaDER 3/85 study for 1/78 sampling.



AR307658

JOB No.:	923-6112	SCALE:	AS SHOWN
DR BY:	WME	DATE:	02/23/94
CHK BY:		FILE No.:	PA17-229
REV BY:		DR SUBTITLE:	01

HISTORICAL FISH TISSUE SAMPLING RESULTS (KEPONE AND MIREX)

RUETGERS-NEASE CORPORATION

FIGURE 10

The National Study of Chemical Residues in Fish (NSCRF)(USEPA 1992) is the most recent nationwide screening investigation to determine the prevalence of selected bioaccumulative chemicals in fish. Mirex was one of the sixty chemical analytes; kepone was not monitored. Between 1967 and 1984, the National Contaminant Biomonitoring Program (NCBP) also monitored the level of organochlorine chemicals in fish. Only the results from the 1976-1979 sampling were available in the literature (U.S.F&WS 1983); while fish samples from the 112-site nationwide program were screened for residues of mirex, no mirex data were included in the 1983 report. There are data from the U.S.EPA program. The 1992 NSCRF study sampled bottom-dwelling (whole body) as well as game fish (fillet) at 314 sites. Mirex was detected at 38% of the sites. The maximum detected levels were 225 ng/g (ppb); the average tissue level was 4 ppb for white sucker (whole body), 15 ppb for catfish (whole body), and 44 ppb for brown trout (fillet). These levels are generally in line with the most recent (mid-1980s) historical fish tissue values in Spring Creek and with the RI fish tissue data (i.e. <25 ppb - 330 ppb).

3. Bioassay Data

Water and sediment bioassays were conducted on samples collected at the Site as part of the RI. These data can be used to address the issue of whether the chemicals of potential concern are bioavailable to the fish and invertebrates of Spring Creek and if they are present at levels that are toxic to some species. The results are summarized below; interpretations within the context of the overall ecological assessment are presented in Section G, Risk Characterization.

a) Water Bioassays

Bioassays were conducted on samples of three water sources that enter Spring Creek in the vicinity of the Ruetgers-Nease facility: (1) Thornton Spring just upstream of its confluence with Spring Creek; (2) ground water treatment effluent from the Ruetgers-Nease facility; and (3) the drainage ditch (which receives water from the ground water treatment facility and from nonpoint source discharges and runoff along its 2,000-foot length) just upstream of its confluence with Spring Creek. Testing was also conducted on water collected from Spring Creek, upstream from the confluences of Thornton Spring and the drainage ditch. Acute and short-term (7-day)

chronic toxicity tests following USEPA protocols were conducted using standard freshwater species. A full report of the study is included as Appendix G of the RI Report.

There was no evidence of acute toxicity (*Daphnia magna and Ceriodaphnia dubia*) for any of the water sources. The results of the chronic toxicity testing are as follows:

- **Thornton Spring** - Concentrations of Thornton Spring water of 60 percent and greater produced reductions in the growth of fathead minnow larvae (*Pimephales promelas*). Reproduction of *Ceriodaphnia dubia* was impaired in 80 percent and higher concentrations of Thornton Spring water.

- **Ground water treatment facility effluent** - Concentrations of effluent at 60 percent and greater produced a reduction in growth of *P. promelas*. Survival and reproduction of *C. dubia* were not affected at any concentration.

- **Drainage ditch** - Survival and growth of *P. promelas* was reduced only at the 100 percent concentration of ditch water. Survival and reproduction of *C. dubia* were not affected at any concentration.

- **Upstream Spring Creek** - Upstream water did not adversely affect *C. dubia* survival or reproduction. *P. promelas* growth was not affected, however, upstream water did adversely affect *P. promelas* survival in one of two tests when compared to survival of fish in dechlorinated tap water controls.

In summary, no acute toxicity was observed during any of the tests. Some effects on survival, reproduction, and/or growth were observed following longer exposures to all four water sources (including the upstream sample) in the 100 percent (undiluted) samples. Reductions in growth and/or reproduction were observed in both the groundwater treatment facility effluent and the Thornton Spring sample concentrations of 60 percent or greater.

b) Sediment Toxicity Bioassays

Bioassays were conducted using *Chironomus tentans* (midge) and *Hyalella azteca* (amphipod) and three sediment samples collected from Spring Creek. The samples were collected: 1) at a point upstream from the confluence of Thornton Spring in the same area as the previous SW3/SED3 sampling for chemical analysis (designated SC-BACKGRD); 2) immediately downstream from the confluence of Thornton Spring (designated SC-TS); and 3) adjacent to Houserville Park which is in the same area as the previous SW2/SED2 sampling (designated SC-Park). The midge and amphipods were exposed to the three sediments for 14 days; amphipods and midge survival and midge growth were assessed.

The tests were conducted in 1993 by the University of Mississippi Biological Field Station (UMBFS). Test methodology conformed to standard American Society of Testing and Materials (ASTM) solid phase toxicity guidelines. Tests were conducted with undiluted and diluted (90%, 80%, 70%, and 60%) sediments. The test methods and results are detailed in Appendix M of the RI Report (*Revised Final Report to the Spring Creek Sediment Testing Program, Weinberg Consulting Group, January 26, 1994*). The results were as follows:

- **SC-Background** - No toxicity was observed. Amphipod and midge survival were not affected by exposure to the sediment; survival was 98 percent for amphipods and 100 percent for midge. Midge growth was not affected; mean dry weight was comparable to that for the reference control group.

- **SC-TS (Thornton Spring)** - Amphipod and midge survival was not affected; they were 95-100 percent and 80-95 percent, respectively. Survival of the control group was 100 percent and 91.4 percent, respectively. Midge growth was (statistically) significantly decreased in the 90 percent treatment. Mean midge weight was also decreased in the 100 percent treatment (undiluted sample), but this decrease was not statistically different from the control. The lack of a statistically significant response in the undiluted sample suggests that the response

in the 90 percent treatment may not be related to chemical concentration, although this cannot be determined definitively.

- **SC-Park** - Amphipod and midge survival was not affected; they were 93-95 percent and 87.5-92.5 percent, respectively. Survival of the control group was 100 percent and 91.4 percent, respectively. Midge growth was (statistically) significantly decreased in each of the 100-, 90-, and 80-percent sediment treatments. The growth reductions were approximately equal, roughly 30 percent, for each of the three treatments, although the mean midge weights for these treatments were within the range of the measured weights for the midge in the control group. Growth was not statistically different from the controls in the 70 percent or 60 percent sediment levels.

4. Ecological Resource Characterization Data

Ecological resources, both habitats and biota, in the study area were characterized using available published literature, consultation with regional/local experts, and a qualitative field survey conducted by SMC as part of the RI. This effort provided a historic as well as a current context in which to prepare the assessment.

In an effort to develop an accurate listing of terrestrial fauna in the area, representatives from PADER, the Pennsylvania State University, and the Pennsylvania Game Commission were consulted (Appendix H). No data on the diversity of terrestrial fauna were available from the sources contacted; however, detailed information on areas within Centre County that are considered key to the maintenance of biological diversity and ecological integrity was prepared by the Western Pennsylvania Conservancy (Appendix I; see pp. 143-151, State College quadrangle).

a) General Habitat Characteristics

The Site and study area lie within the Spring Creek basin in south-central Centre County, Pennsylvania, and within the Appalachian Mountain Section of the Valley and Ridge Physiographic Province (PADER 1990). The region is characterized by a series of long, high ridges and broad valleys primarily composed of limestone and dolomite with relatively minor shale and sandstone formations. Soil associations in the region were formed in

residual and colluvial material weathered from these formations. Because of the fractured carbonate geology of the area, open solution channels and sinkholes are relatively common in the region.

The Spring Creek basin drains approximately 144 square miles of rural land before flowing into Bald Eagle Creek at Milesburg. The average channel slope for Spring Creek is 22.1 feet per mile and the creek is approximately 25 miles long (PADER 1990). The watershed has a maximum elevation of 2,410 feet above mean sea level at the Little Flat lookout tower on Tussey Mountain and a minimum elevation of 690 feet at the mouth. Ground water-surface water connections are common and some of the emerging springs support very large flows (i.e., up to 8,000 gpm) (PADER 1989). Expansive wetlands are not common in the region, although several small wetlands, less than 5 acres in size, are found in association with drainageways (riparian wetlands) or as isolated depressions.

Open space and agriculture dominate the Spring Creek basin and much of the open space is contained within publicly owned lands. However, residential, commercial, and light industrial land uses have increased greatly in the past decade, particularly near the borough of State College. Limestone and dolomite mining is another significant land use activity within the watershed (PADER 1990), and the infrastructure of roadways and municipal waste water treatment plants needed to support these land uses has grown proportionally. In light of advancing development, county and regional planning commissions have been working with the Pennsylvania Department of Transportation and other state agencies to minimize development impacts to the Spring Creek basin.

White pine and hemlock climax forests dominated the Spring Creek basin prior to 1900 but were lost to timbering, farming, fire, and the chestnut blight (PADER 1990). Today, second growth stands of oak and various other hardwoods dominate the higher elevations, while evergreens and rhododendron grow along the streambanks and in the valleys.

b) RI Description of RMU Habitats and Biota

The following descriptions of the habitats and biota in the study area come from the qualitative field survey conducted by SMC as part of the RI.

The descriptions are broken out by the individual RMUs in order to facilitate the assessment.

1) **RMU1**

RMU1 is an approximately 15-acre grassy, upland field that is maintained by periodic mowing. This area is bordered by fenced portions of the Ruetgers-Nease facility and Struble Road to the northeast, First Avenue to the southeast, a forested swale to the southwest, and the drainage ditch and concrete-lined pond to the northwest. Private residences are found along the south side of First Avenue and open meadows are common adjacent to these homes and upgradient from the facility. Wetlands are not found at RMU1. Based on the open field habitat, terrestrial animals believed to be present at RMU1 would include terrestrial invertebrates (e.g., earthworm), snakes, fieldbirds (e.g., red-winged blackbird), small mammals (e.g., shrews, voles, and moles), and possibly raptors (e.g., red-tailed hawk).

2) **RMU2**

RMU2A covers the area of the drainage ditch on the facility. RMU2B covers the off-site drainage ditch as it leaves the facility property, parallels the access roadway southwest of the facility, runs along Pennsylvania Route 26 (a light industry area), to its confluence with Spring Creek. This total distance is approximately 2,000 feet. This ditch, part of which runs adjacent to the Conrail railroad tracks on the facility, is characterized as an intermittent drainageway with minimal bankside vegetation.

Sediments in on-site portions of the ditch are sands and silts, while the ditch in the downstream section adjacent to Route 26 is alternately composed of unconsolidated cobble, sand, and exposed bedrock. Flow in the drainage ditch is dependent primarily upon the discharge from the Ruetgers-Nease ground water treatment facility. On and off site, the flow is dependent upon storm water runoff. The ditch sediment was saturated at the time of the field inspection, but there was no flowing water.

Organisms that spend significant amounts of time in or on the ditch sediments likely include earthworms, snails, and midge larvae. Green frogs and red-winged blackbirds were observed along the banks of the ditch. Judging by the tracks in the ditch sediments, raccoons are present in RMU2.

3) RMU3

RMU3, Thornton Spring, is a perennial first-order stream that originates from a ground water seep at the southern end of Nittany Mountain. Thornton Spring flows approximately 200 feet past a private residence before emptying into Spring Creek through a culvert under Pike Street just south of Pennsylvania Route 26. The streambed of Thornton Spring is 2-4 feet wide and is comprised of unconsolidated sand, gravel, and cobble, and it contains relatively little organic carbon. Flow rate in Thornton Spring varies substantially with rainfall, but is estimated to be approximately 0.75 cfs under normal conditions. This flow rate estimate represents approximately one percent of the flow rate of Spring Creek at its confluence with Thornton Spring. Land immediately adjacent to Thornton Spring contains a hardwood forest (i.e., maple, oak, walnut, ash, and cherry), a few shrubs (i.e., raspberry and forsythia), and a residential lawn that borders the stream before it enters the Pike Street culvert and empties into Spring Creek.

Midge larvae, earthworms, and crayfish were found in the stream sediments. Small mammals, amphibians, and reptiles typically found in association with this type of forested stream, but which were not observed at RMU3, include the white-footed mouse, opossum, raccoon, woodpeckers, frogs, salamander, and garter snake.

4) RMU4

RMU4 includes Spring Creek and its riparian zone in the vicinity of the Pike Street Bridge. This location is approximately 0.75 miles upstream from Thornton Spring. Spring Creek is a third-order cold water stream with a riparian zone that is alternately forested and maintained as residential lawns. The canopy over the creek covers 30-40 percent of the stream. Sediments are composed primarily of sand,

gravel, cobble, and a substantial amount of particulate organic matter (i.e., leaf packs and woody debris) is also present. At the Pike Street Bridge, the flow rate under normal hydrologic conditions is estimated to range from 50 to 80 cfs.

Spring Creek in the vicinity of RMU4 has a rich benthic macroinvertebrate community consisting primarily of aquatic insects. Together with amphibians and reptiles commonly found along the stream bank (i.e., green frog, painted turtle, and water snake), stream invertebrates support a relatively large forage and game fish population. Appendix J provides a listing of aquatic invertebrates and fish species found in Spring Creek. Muskrats, raccoons, dippers, mallards, great blue heron and several songbird species (i.e., cedar waxwing and warbler) were also noted during the RI survey.

5) RMU5

RMU5 includes a section of Spring Creek that flows adjacent to Houserville Park, approximately 1 mile downstream from the confluence of Thornton Spring and below the confluence of the drainage ditch near Pennsylvania Route 26. This stream segment is physically similar to RMU4 but, on a qualitative level, fine sediments (especially sands) seem to make up a greater percentage of the substrate. Additionally, the extent of mature riparian vegetation at the Houserville Park location is considerably less than that upstream, thus shading is reduced. Houserville Park is a maintained recreational area with ballfields that extend all the way to the east bank of Spring Creek. Land adjacent to the west bank of Spring Creek (opposite Houserville Park) was under development at the time of the field inspection. In this open space, a new roadway was cut and earth moving and other preparations for building were occurring. An operating livestock farm is located immediately downstream from Houserville Park. Livestock are restricted from access to Spring Creek by fencing.

With the exception of Canada geese, which were observed at Houserville Park, biota are the same as these described for RMU4 (see Appendix J for aquatic invertebrate and fish species found in Spring Creek).

6) **RMU6**

Spring Creek in the vicinity of the Benner Spring Fish Hatchery is an approximately fourth-order stream segment located about 4 miles downstream from RMU5. The stream at this point is formed in conjunction with Slab Cabin Run (confluence above RMU6), which also carries waste water discharges from the Pennsylvania State University municipal waste water treatment plant. Spring Creek is wider and deeper at RMU6 than upstream; flow rate is estimated at 80 to 120 cfs, substrates are dominated by cobbles (riffles) and sand/silt (pools), and macrophytes are sparsely distributed. Various shrubs, vines, and wildflowers are common along the edge of the creek, and most of the riparian area is forested with mixed deciduous and coniferous trees. Additionally, the streambed is contained within a distinct valley at RMU6 and there are very few wetland habitats adjacent to the stream.

Biota at RMU6 are similar to those species identified at RMU4 and RMU5 (see Appendix J for aquatic invertebrate and fish species found in Spring Creek).

c) **Species of Special Concern**

For the Spring Creek watershed upstream from Bellefonte (this includes the entire Site), there are 36 plants and animals listed as "Species of Special Concern" by the Pennsylvania Natural Diversity Inventory (PNDI) (PADER 1991). The PNDI listing is inclusive of all federally listed rare, threatened, or endangered species. Of the 36 species of special concern, 4 terrestrial plants are confirmed to be present within 5 miles of State College (PADER 1991). These include Geyer's Sedge (*Carex geveri*, endangered), lupine (*Lupinus perennis*, rare), low serviceberry (*Amelanchier humilis*, tentatively undetermined), and gay-feather (*Liatris scariosa varnieuwlandii*, tentatively undetermined). No animal species of special concern were identified by PNDI as residing within 5 miles of State College, which includes the entire Site.

d) Historical Biological Community Data

With the exception of fish collected as part of tissue residue analysis, biota were not collected from Spring Creek as part of the RI. Data available from the published literature and particularly from Pennsylvania Department of Environmental Resources (PADER) and the Pennsylvania Fish Commission (PFC) on the condition of the fishery (i.e., brown trout) and benthic macroinvertebrates in Spring Creek were collected and evaluated to provide both a recent and historical context within which to evaluate the current condition. It should be noted that although other fish species inhabit Spring Creek, no readily available data on population sizes or conditions of these other species exists. Nevertheless, information on brown trout populations in Spring Creek should provide a reasonable upper-bound estimate of potential impacts on fish populations inhabiting the creek, as this species can be regarded as a reasonable sentinel species for the evaluation of the health of Spring Creek. This is because trout: 1) are at the top of the aquatic food chain, and therefore are potentially exposed to bioaccumulative chemcials to the greatest extent; 2) are members of a family (Salmonidae) that is often very sensitive to chemical toxicants; and 3) are an ecological resource that is highly valued from a fishery standpoint.

1) Fish (trout) Populations

The earliest available information on fish populations in Spring Creek is from the late 1950s when the PFC began reporting on the impacts of high organic loading from sewage treatment plants in the area (PFC 1959). Impacts (recorded as: an oil slick; chemical odor; and the presence of sewage tolerant benthic species) were reported from the area where Slab Cabin Run enters Spring Creek down to as far as the Benner Spring Research Station, several miles downstream, and even further downstream near Bellefonte. Trout population size and productivity were not reported as part of this study.

Carline (1991) reports that from 1958 to 1967, the wild brown trout population in Spring Creek ranged in size from about 185 to 420 adults/hectare and in biomass from about 57 to 123 kg/hectare. Abundance and biomass were approximately thirty to fifty percent lower, respectively, for fish collected from the upper reaches of Spring

Creek (i.e., one to two miles above the route Route 26 bridge, in the Lemont area) versus the lower reaches of Spring Creek (i.e., ten to eleven miles downstream of the Route 26 bridge, in the area upstream of Bellefonte and the confluence of Buffalo Run).

A 1981 PFC report (PFC 1981) stated that the fish species composition in Spring Creek had remained stable for the ten years preceding the publication. Estimated standing stocks of brown trout ranged from 31 kg/hectare to 146 kg/hectare and were said to be typical of a fertile limestone stream except for the area impacted by two sewage treatment plants. The report also addressed numerous spills of fuel oil and gasoline, the largest of which resulted in a fishkill downstream of where Route 26 crosses Spring Creek (PFC 1981).

Stocking of hatchery trout was suspended downstream of the Site in 1978. In 1982, all stocking was suspended and a no-harvest advisory was put into effect for a 19-mile stretch of Spring Creek going from about four miles above the Site to about 15 miles below the Site. Since that time, the size of the wild trout population in Spring Creek has increased. From 1980 to 1988, the mean overall number of adult fish (the most closely monitored group) in this area increased from 345 to 725 fish/hectare (210% increase) and biomass of adult fish rose from 93 to 143 kg/hectare (154% increase) (PFC 1981, PADER 1990 and Carline 1991).

A 1990 PADER report (PADER 1990) stated that Spring Creek supports an excellent brown trout population, with the sparsest numbers downstream of the University Area Joint Authority (UAJA) treatment plant (about 4-5 miles downstream of the Site). PADER notes that there are currently twelve NPDES-permitted discharges of treated wastewater into Spring Creek. Siltation has resulted in limited suitable spawning areas and decreased fish egg survival between Slab Cabin Run and the Benner Spring hatchery. Trout Unlimited had initiated a stream bank stabilization program in an effort to improve this section of the creek.

Table 61 provides a summary of the historical data on brown trout abundance and productivity measures in Spring Creek. Figures 11 and 12 locate the historical sampling points on a map of Spring

Creek. These data suggest that Spring Creek supports a reproducing wild trout population and that fishing and stocking pressures were the main historical checks on fish density in Spring Creek. The PFC has set a minimum criteria of 40 kg/hectare for Class A Wild Trout Waters. Spring Creek consistently meets this criteria with a mean overall historical wild brown trout biomass ranging from a low of 57 kg/hectare (1958) to a high of 143 kg/hectare (1988). The 1990 PADER report concludes that "Spring Creek supports an excellent brown trout population ...".

The most recent sampling data (1988) provides the most complete data with which to compare trout population characteristics along the 19-mile reach of Spring Creek. With exceptions of two sampling sites, roughly three to eight miles below the Route 26 bridge (i.e. near and below the Benner Spring Hatchery), the numbers of adult brown trout are roughly equal. Trout biomass decreases somewhat in the area three to five miles below the Route 26 bridge and is at its highest at the furthest downstream stations, between eight and fifteen miles below the Route 26 bridge.

2) Benthic Macroinvertebrate Populations

Historical data available from benthic macroinvertebrate studies within the Spring Creek watershed have been described in the Remedial Investigation revisions pages 1-40.43 to 1-40.58 (October 1993). The first study of the benthic community was conducted by the PFC in 1959 and a total of twelve studies have been conducted subsequently by PADER between 1971 and 1992. A summary of the benthic macroinvertebrate data collected during these studies is presented in Figure 13, which also shows the locations of the samples within the Spring Creek watershed relative to the Site.

Figure 13 summarizes historical abundance data for five benthic taxa that inhabit Spring Creek. Although other taxa inhabit Spring Creek and have been identified in past studies (see Table 71 in Section

TABLE 61
Historical Wild Fish Population Estimates in Spring Creek Based on Sampling Data

Stream Miles from Site (approx.)	Sampling Year	Number of Juvenile and Adult Fish/Hectare	Kg of Juvenile and Adult Fish/Hectare	Number of Adult Fish/Hectare	Kg of Adult Fish/Hectare	Reference
2-4 upstream	1981	2,395	31	--	--	PFC 1981
	1988	--	--	861	130	PADER 1990
0-2 upstream	1958	--	--	267	57	Carline 1991
	1965	--	--	185	66	Carline 1991
	1966	--	--	358	110	Carline 1991
	1967	--	--	420	123	Carline 1991
	1980	668	--	310	56	Carline 1991
	1981	1,114	59	--	--	PFC 1981
	1988	1,343	--	927	135	Carline 1991
0-1	1980	531	--	453	116	Carline 1991
downstream	1988	851	--	678	104	Carline 1991
1-3	1981	513	131	--	--	PFC 1981
downstream	1988	--	--	799	138	PADER 1990
3-5	1980	386	--	321	145	Carline 1991
downstream	1981	310	120	--	--	PFC 1981
	1988	806	--	373	89	Carline 1991
5-8	1981	2,509	146	--	--	PFC 1981
downstream	1988	--	--	310	118	PADER 1990

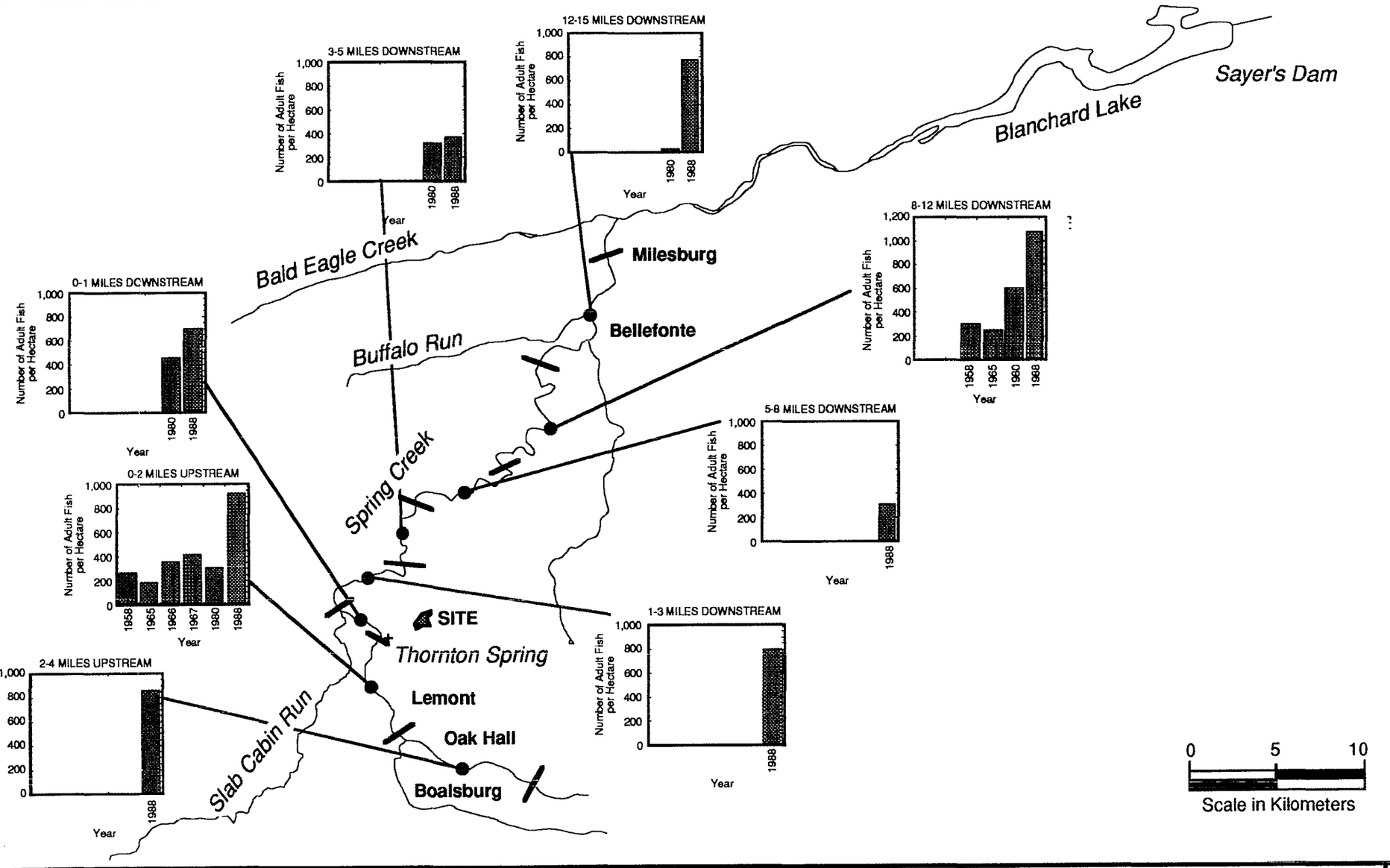
TABLE 61
Historical Wild Fish Population Estimates in Spring Creek Based on Sampling Data

Stream Miles from Site (approx.)	Sampling Year	Number of Juvenile and Adult Fish/Hectare	Kg of Juvenile and Adult Fish/Hectare	Number of Adult Fish/Hectare	Kg of Adult Fish/Hectare	Reference
8-12	1966	--	--	306	78	Carline 1991
downstream	1967	--	--	252	76	Carline 1991
	1980	987	--	609	136	Carline 1991
	1981[1]					PFC 1981
	1988	1,568	--	1,077	238	PADER 1990/ Carline 1991
12-15	1980	46	--	31	11	Carline 1991
downstream	1988	983	--	778	188	Carline 1991
Mean	1958	--	--	267	57	Not applicable
Upstream	1965	--	--	185	66	Not applicable
	1966	--	--	358	110	Not applicable
	1967	--	--	420	123	Not applicable
	1980	668	--	310	56	Not applicable
	1981	1,755	45	--	--	Not applicable
	1988	1,343	--	894	133	Not applicable
Mean	1966	--	--	306	78	Not applicable
Downstream	1967	--	--	252	76	Not applicable
	1980	488	--	354	102	Not applicable
	1981	1,110	132	--	--	Not applicable
	1988	1,052	--	669	146	Not applicable

TABLE 61
Historical Wild Fish Population Estimates in Spring Creek Based on Sampling Data

Stream Miles from Site (approx.)	Sampling Year	Number of Juvenile and Adult Fish/Hectare	Kg of Juvenile and Adult Fish/Hectare	Number of Adult Fish/Hectare	Kg of Adult Fish/Hectare	Reference
Mean	1958	--	--	267	57	Not applicable
Overall	1965	--	--	185	66	Not applicable
	1966	--	--	332	94	Not applicable
	1967	--	--	336	100	Not applicable
	1980	524	--	345	93	Not applicable
	1981	1,368	97	--	--	Not applicable
	1988	1,175	--	725	143	Not applicable

NOTES:
-- Indicates no data availabe
[1] Section contained so few fish that population estimates could not be made.



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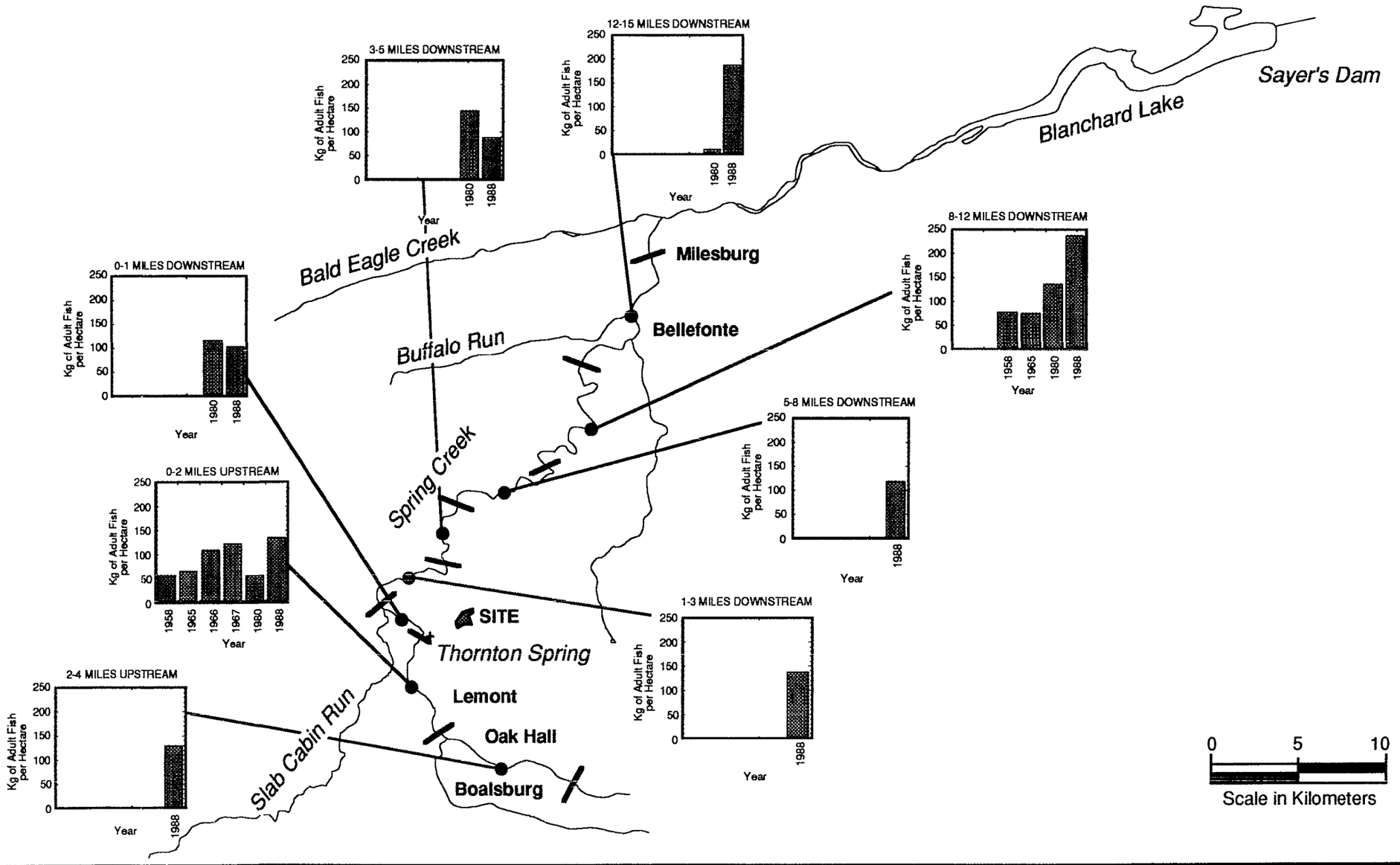
NUMBERS OF ADULT BROWN TROUT PER HECTARE ALONG SPRING CREEK

Figure 11



KILOGRAMS (Kg) OF ADULT BROWN TROUT PER HECTARE ALONG SPRING CREEK

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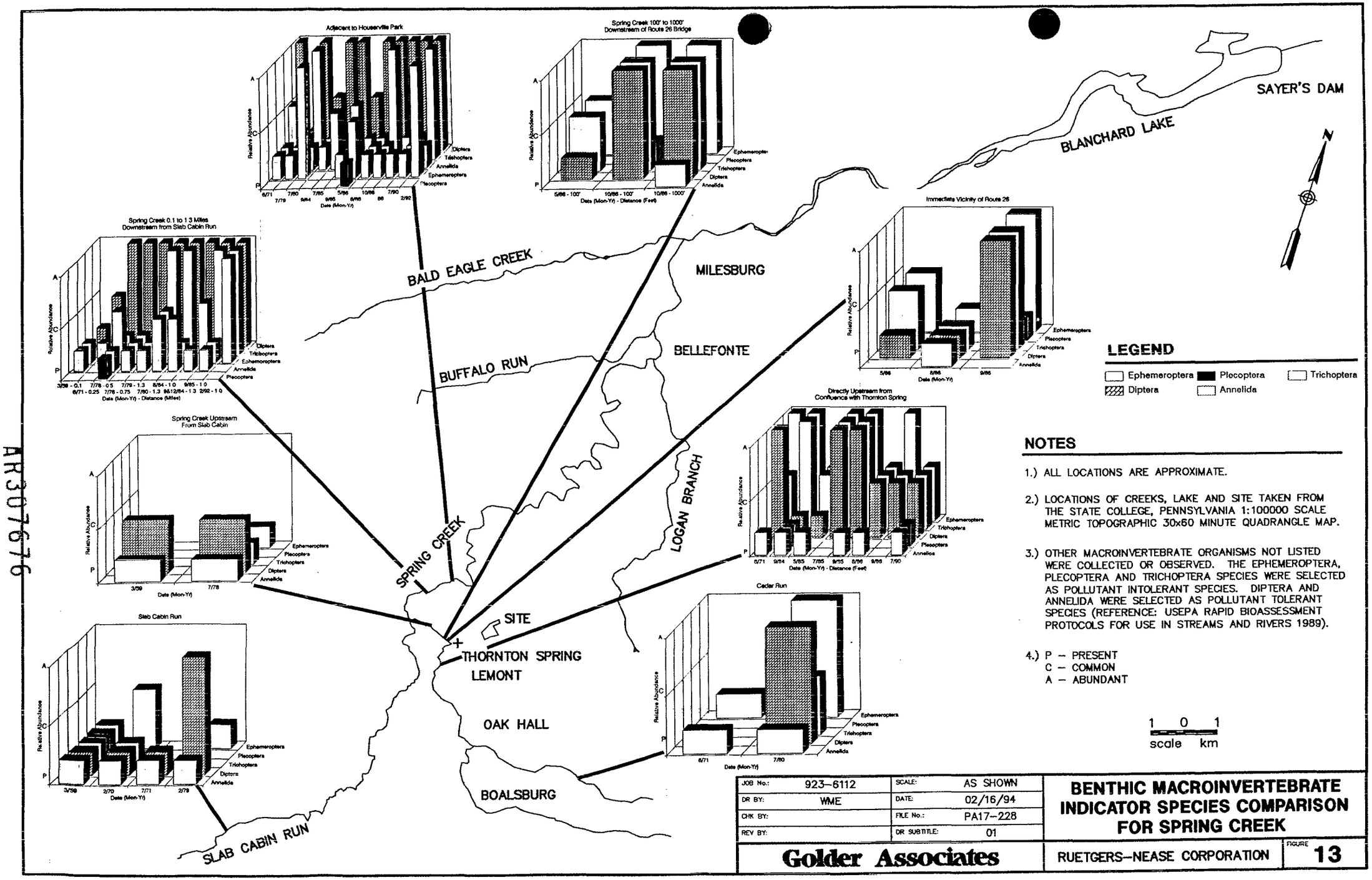
Figure 12



AR307616

JOB No.:	923-6112	SCALE:	AS SHOWN
DR BY:	WME	DATE:	02/16/94
CHK BY:		FILE No.:	PA17-228
REV BY:		DR SUBTITLE:	01

Golder Associates

BENTHIC MACROINVERTEBRATE INDICATOR SPECIES COMPARISON FOR SPRING CREEK

RUETGERS-NEASE CORPORATION

FIGURE 13

F), the taxa presented on Figure 13 are used here as indicator benthic groups to facilitate comparisons of species composition throughout Spring Creek. These indicator groups were selected using the concept of "polution tolerant" and "Pollution intolerant species" outlined by the USEPA in the Macroinvertebrate Field and Laboratory Methods for Evaluating the Biological Integrity of Surface Waters (EPA/600/4-90/030) and in the EPT index according to the Rapid Bioassessment Protocols for Use in Streams and Rivers, Benthic Macroinvertebrates and Fish (EPA/444/4-89-001). Individual taxa were classified by the USEPA in these protocols on the basis of their established tolerance or intolerance to various levels of pollutants or contaminants from documented field studies. Intolerant species are generally not found in water with organic contaminants and are intolerant of even moderate reductions of dissolved oxygen. Tolerant species are capable of thriving under almost anaerobic conditions. Since pollutant tolerant organisms can thrive in both healthy or degraded aquatic habitats, they are not indicators of significance. However, the presence of pollutant intolerant species provides evidence of healthy aquatic conditions.

Pollutant intolerant species, as identified by the USEPA, are Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) collectively referred to as "EPT." Pollutant tolerant benthos include Diperta and Annelida species (e.g., Chironomidae and Oligochaetes).

The results of the benthic macroinvertebrate studies in Spring Creek exhibit both good-to-excellent, and occasionally impaired, water quality and aquatic life. The results reflect that while impacts to the benthic communities exist, they are sporadic and do not appear to be attributable to a single source. In general, areas downstream of Thornton Spring, though periodically impacted, reflect that several factors (e.g., Slab Cabin Run water quality) have potentially caused adverse impacts to Spring Creek historically. In Figure 13 the area upstream of the confluence with Slab Cabin Run appears to reflect impacts to Spring Creek during the only two sampling events (1959 and 1978) conducted in this portion of the stream. However, the March 1959 survey concluded impacts noted in Spring Creek were attributable

to the Pennsylvania State University Sewage Treatment Plant discharge and that areas upstream of the confluence with Slab Cabin Run were classified as a "clean water zone." In addition, the July 1978 survey was conducted as part of an on-going study to evaluate impacts suspected within the vicinity of Thornton Spring. The study did not demonstrate depression of the macroinvertebrate community below Thornton Spring; specifically, the presence of Ephemeroptera and Trichoptera species during the 1978 survey reflects a generally healthy aquatic environment within the stream. Putting year-to-year and seasonal variations aside, the most recent data (1985-1992) reflect that there are both pollutant tolerant and pollutant intolerant species in abundance at sampling locations above and below the Site.

C. Selection of Chemicals for Evaluation

Based on the RI characterization analytical data, past operations at the facility, and the availability of environmental persistence and ecotoxicological effects data for chemical substances that were detected, the Site-related chemicals of concern are limited to mirex and kepone. These were the only two chemicals which were detected in all of the media sampled including soils, sediment, surface water, and fish. Their environmental fate and toxicological profiles are presented below.

The volatile organic compounds (VOCs) that were detected at the Site are not addressed individually in the risk assessment. They are, however, factored into the assessment because: (1) to the extent that they are in the water and sediments, they would have been present in the bioassay tests conducted with samples taken from the study area; and (2) the historical and RI fish and benthic macroinvertebrate community data would reflect exposure to VOCs, to the extent that they are present in the sampling areas. Because VOCs are not readily accumulated in tissues, they are not candidates for foodchain exposure estimation in wildlife. Some discussion about the environmental fate and toxicity of VOCs follows.

Twenty-seven volatile organics (VOCs) were detected in either surface water, sediment and/or surface soil samples (Appendix O, Table O-8). These chemicals are considerably less persistent than mirex and kepone in any environmental matrix. To the extent that data exist, VOCs have been shown to degrade in both water and soil systems, and they are not expected to bioaccumulate to any significant degree. Some of the VOCs have been shown to be toxic to sensitive aquatic species; however, ecotoxicological data and

thresholds are not available for many of the 27 chemicals detected at the Site. Although there may be some toxicity associated with VOCs in the drainage ditch and Thornton Spring, and this may explain the toxicity to fish and daphnids observed in the laboratory aquatic toxicity tests, those same tests indicate that there would not be any toxic effect once these sources are diluted as they enter Spring Creek.

The only VOCs detected in Spring Creek were toluene and 1,1,2,2-tetrachlorethane in the water, and acetone, 2-butanone, and toluene in sediments. The maximum measured concentrations of toluene and 1,1,2,2 tetrachlorethane in Spring Creek water were 1 and 3 μg/L, respectively (Appendix O, Table O-8). These levels are not of ecological concern given their lowest observed effect levels (LOELs) are 17.5 mg/L (aquatic acute) and 2.4 mg/L (aquatic chronic), respectively (U.S.EPA 1986d). The maximum measured concentrations of acetone, 2-butanone and toluene in Spring Creek sediments were 110, 3, and 9 μg/kg, respectively (Appendix O, Table O-8). These levels are not of ecological concern based on estimated sediment toxicity threshold levels derived using equilibrium partitioning equations in the U.S.EPA Interim Sediment Criteria Document (1988c). The estimated sediment thresholds are greater than 1,000 μg/kg for acetone and toluene and greater than 500 μg/kg for 2-butanone. The VOCs were not assessed further for ecological risks as part of this assessment.

1. Mirex

a) Properties and Status

Empirical formula:	C_{10} Cl_{12}
Molecular weight:	545.5
Water solubility:	1 ppb (freshwater)
Henry's Law constant:	5.16 x 10^{-4} atm-m^3/mol
Log K_{ow}:	6.89 (Veith et al 1979)
K_{oc}:	2.4 x 10^7 (HSDB 1992)

Mirex (perchloropentacyclodecane) has been used extensively in pesticidal formulations to control the imported fire ant and as a flame retardant in electronic components, plastics, and fabrics. In 1978, USEPA banned the use of mirex as a pesticide, partly because of the hazards it imposed on non-target biota. These hazards included delayed mortality in aquatic and terrestrial fauna, adverse effects on reproduction, early growth, and

development, plus high bioaccumulation and biomagnification in the foodchain (Eisler 1985). The U.S. Food and Drug Administration (FDA) has set an action level of 0.1 ppm mirex in fish for human consumption. Fish that contain mirex above this level are not considered safe in commerce (43 FR 14736).

b) Fate

Mirex is a very stable and persistent organochlorine compound, being resistant to chemical, photolytic, microbial, and thermal degradation. There is evidence, however, for some degradation to monohydro- (photomirex) and dihydro-derivatives, which have biological activities similar to mirex (Eisler 1985). Mirex adsorbs very little UV light in the environmentally relevant range of >290 nm. A photodegradation experiment conducted in pure water for 6 months showed a half-life (i.e., time required for half of the starting material to be lost) of about 1 year (Smith 1978).

Mirex has low solubility in water, not exceeding 1 ppb in freshwater and 0.2 ppb in seawater. It is highly soluble in fat and accumulates in fatty tissue (log K_{ow} of 6.89). Mirex is rapidly adsorbed onto various organic particles in the water column, including algae, and eventually is removed to the sediments. With its relatively high K_{oc} value of 2.4×10^7, mirex will strongly adsorb to organic materials in soil and will be immobile except for movement via erosion to surface waters (Smith 1978). Mirex is persistent in terrestrial and aquatic soils/sediments. Degradation half-life estimates range to 10 years or more. In biological systems, the elimination half-lives range from 30 days in quail to 130 days in fish, and to more than 10 months in rats (Eisler 1985). The low Henry's Law constant (5.16×10^{-4} atm-m^3/mol) suggests that mirex is unlikely to volatilize rapidly from soil or surface water.

c) Predicted Bioaccumulation

Mirex bioaccumulates in aquatic organisms, with bioconcentration factors (BCFs) in the thousands for algae and aquatic invertebrates, and up to tens of thousands or more for fish. A log BCF of 7 was calculated for mirex in Lake Ontario rainbow trout (Oliver and Niimi 1985). Bioaccumulation factors (BAFs) for birds and mammals exposed to mirex in the foodchain are generally less than 50 (Eisler 1985). The highest levels of mirex in exposed

organisms are present in fat and in eggs which are high in lipids. [NOTE: BCF applies to aquatic or terrestrial organisms and is based on total chemical uptake from all relevant exposure pathways, but particularly via food.]

No data were available for bioaccumulation in invertebrates exposed to mirex in soils or sediments. Therefore, in this assessment, it is necessary to estimate accumulation using a model.

Based on general fugacity concepts, Connell and Markwell (1990) have suggested that bioaccumulation of organic compounds in invertebrates can best be described as a three compartment model involving the sediment/soil, interstitial water, and the biota, in which the partitioning to organisms from water (BAF_w) divided by the partitioning from water to soil/sediment (K_p) approximates the bioaccumulation from soil/sediment to invertebrate. Partitioning of organic compounds between water and sediment/soil is a function of the compound's affinity for soil/sediment organic carbon (usually expressed as the K_{oc} partition coefficient) and the amount of organic carbon in the sediment/soil (expressed as % carbon). Partitioning of organic compounds from water to organisms (actually organism lipid) is a non-linear function of the affinity of a compound for organic material relative to water (usually expressed as the octanol:water partition coefficient or K_{ow}) and the lipid content of the organism (expressed as % lipid).

Connell and Markwell's equation describing the bioaccumulation of a compound from soil/sediment to invertebrate (BAFs) is as follows:

$$BAF_s = \frac{BAF_w}{K_p} = \frac{(\%\ \text{lipid})(K_{ow})^a}{(\%\ \text{Carbon})K_{oc}}$$

where: **a** is the non-linearity constant for bioaccumulation from water.

This fugacity equation yields a soil-to-earthworm BAF for mirex of 0.51 using the following values:

(1) K_{ow} 7,762,471 from log K_{ow} of 6.89;

(2) K_{oc} of 24,000,000;

(3) typical earthworm lipid content of 0.85 percent (Rao and Davidson 1980);

(4) assumed Site-area soil organic carbon of 5 percent (Manahan 1991); and

(5) a non-linearity constant (a) for organochlorine compound accumulation from water to earthworms of 1.14 (Lord et al. 1980).

This 0.51 BAF value is similar to the measured average BAF (0.56) for earthworms exposed to soils containing the similarly organophilic compound TCDD (log K_{ow} 6.9) at soil concentrations ranging from 500 to 5,000 μg/kg (Reinecke and Nash 1984).

The same fugacity equation yields a sediment/to/benthic macroinvertebrate BAF for mirex of 6.95 using the following values:

(1) log K_{ow} of 6.89 (K_{ow} 7,762,471);

(2) K_{oc} of 24,000,000:

(3) average invertebrate insect lipid content of 15 percent (Hanson et al. 1985);

(4) measured sediment organic carbon of 4 percent; and

(5) a non-linearity constant (a) for organochlorine compound accumulation from water to aquatic invertebrates of 1.11 (Markwell et al. 1989).

This 6.95 BAF value for benthic macroinvertebrates is comparable to the measured BAFs for chironomid midge larvae exposed to the structurally similar compound kepone (BAF between 3 and 20 for sediments with 12.3 and

1.5% organic carbon, respectively) in a flow-through laboratory system (Adams 1987).

d) Toxicity (Nonhuman)

In short-term (LC_{50}) studies, aquatic organisms are relatively resistant to mirex toxicity. Delayed mortality, however, has been observed in aquatic species after extended periods of exposure in water (Eisler 1985). This delayed toxicity presumably results from the time required for mirex to accumulate to toxic levels. Delayed mortality was observed for freshwater and estuarine crustaceans (i.e., crayfish and shrimp) after exposures as low as 0.0001 mg/L in the water (USEPA 1986d). [Note, however, that crayfish were observed in Thornton Spring which had an RME concentration of 0.0005 mg/L in the water.] The maximum acceptable toxicant concentration (MATC) determined for sublethal effects is less than 0.0024 mg/L for amphipods based on growth inhibition, less than 0.005 mg/L for bluegills based on growth, and 0.034 mg/L for fathead minnows based on impaired reproduction and emergence, respectively. Other sublethal effects have been observed for algae, invertebrates, and fish species (USEPA 1986d). There are no reported studies that relate toxicity in aquatic species to levels of mirex in their food or in their own tissues as a result of bioaccumulation.

Birds appear comparatively resistant to mirex. Eisler (1985) concludes in his review that "most investigators agree that comparatively high dietary concentrations of mirex had little effect on growth, survival, reproduction and behavior of nonraptors, including chickens, mallards, quail and red-winged blackbirds." In birds, mortality has generally been reported following repeated exposures at high dietary concentrations (Eisler 1985). For example, 27 percent of mallard ducks died when exposed to 100 mg/kg mirex in the diet for 25 weeks, 50 percent of ring-necked pheasants died when exposed to 1,500 mg/kg in the diet for 5 days, and 20 percent of Japanese quail died in 5 days when exposed to 5,000 mg/kg mirex in the diet. Some, but not all, investigators have reported reductions in egg hatchability and chick survival following mirex exposure. A summary of repeat-dose toxicity studies of mirex in birds is presented in Appendix K.

In studies of mammalian species, mirex has been shown to cause decreased weight gain, liver effects, reproductive impairment, and, at

sufficiently high dose levels, mortality. Reported findings from repeat-dose toxicity studies in mammals are summarized in Appendix L.

2. Kepone

a) Properties and Status

Empirical formula:	$C_{10}Cl_{10}O$
Molecular weight:	490.6
Water solubility:	2,000-3,000 ppb (Sax 1984)
Henry's Law constant:	2.50×10^{-8} atm-m^3/mol (Howard 1991)
Log K_{ow}:	4.50 (Howard 1991)
K_{oc}:	$2.4 - 2.6 \times 10^3$ (HSDB 1992)
K_{sw}:	5×10^3 (estimated) (Strobel et al. 1981)

Kepone (chlordecone) has been used primarily as an insecticide bait, especially in ant and cockroach traps. It has also been used effectively against leaf-cutting insects, as a fly larvicide, and as a fungicide against apple scab and powdery mildew. Other insecticide uses include control of the Colorado potato beetle, rust mite on non-bearing citrus trees, and potato and tobacco wireworm on gladioli and other plants (WHO 1984). Kepone is also the base material in the manufacture of another insecticide, Kevelan. In 1976, USEPA banned the use of kepone because of its potential for bioaccumulation in aquatic foodchains and its adverse effects on aquatic and terrestrial organisms (WHO 1990). The U.S. Food and Drug Administration (FDA) has set an action level of 0.3 ppm kepone in fish for human consumption. Fish that contain kepone above this level are not considered safe in commerce (43 FR 14736).

b) Fate

Kepone is not a naturally occurring substance; therefore, its release to the environment will be a result of its manufacture and use as an insecticide and as a degradation product of mirex. Like mirex, kepone is an environmentally persistent compound. Because of its relatively low water solubility and low vapor pressure, the most likely fate for kepone is either transport or storage in soil, sediment, or biological media.

If kepone is released to soil, it will be expected to adsorb to soils, however, some leaching to ground water may occur, especially in sandy soils and soils with low organic matter contents. Kepone's K_{oc} value of 2.4 - 2.6 x 10^3 indicates that kepone will be only slightly mobile in soil (Howard 1991). Biodegradation and hydrolysis are not expected to be important fate processes for kepone in soil (HSDB 1992).

Since kepone has relatively low water solubility (2,000-3,000 ppb), it is not expected that direct transport in solution will be a major distribution pathway. However, kepone released to water is expected to adsorb to sediments (log K_{ow} of 4.5). Strobel et al. (1981) evaluated the sediment/water partition coefficient (K_{sw}) for kepone through field studies of the James River Estuary in Virginia. The K_{sw} was established to be 5 x 10^3, which means that there is a 5,000 to 1 partitioning of kepone in sediments versus kepone in solution. Sediments represent the major reservoir and sink for kepone in the aquatic environment. Kepone is not expected to hydrolyze, biodegrade, or evaporate from water. A half-life of 3.8 to 46 years has been predicted for evaporation from a river one meter deep, flowing at 1 meter per second with a wind velocity of 3 meters per second (HSDB 1992). No data were available concerning the photodegradation of kepone irradiated at environmentally significant wavelengths, but no significant photodegradation in water is expected (HSDB 1992).

The transport of kepone in the atmosphere is not expected to be significant. Because of its low vapor pressure ($<3 \times 10^{-7}$ mm Hg at 25°C), kepone will not readily volatilize in the atmosphere. It is also not expected to directly photodegrade or react with photochemically produced hydroxyl radicals or ozone (HSDB 1992).

c) Predicted Bioaccumulation

Bioaccumulation of kepone occurs in aquatic and terrestrial organisms. The fate of kepone in biological systems depends on the organism involved and the route of exposure. The predominant exposure pathway for organisms in the foodchain system is the aquatic sediment pathway. Kepone tends to accumulate through the foodchain; however, it is eliminated more rapidly and does not partition to the same degree in fat tissue compared to most other polychlorinated xenobiotics.

BCFs for kepone are in the hundreds for algae (*Chlorococcum* 800x; *Nitzschia* 410x; *Thalassisiva* 520x), and well over 1,000 for some invertebrates (grass shrimp 698-13,473) and fish species (fathead minnow 1,100-51,300; bluegill 122-1,821; channel catfish 288-1,163) (HSDB 1992; Roberts and Bendl 1982; Huckins et al. 1982; USEPA 1978; Bahner et al. 1977). Because kepone readily partitions into fish tissue, there is some concern that small mammals and birds that consume fish will also accumulate kepone to toxic levels.

Data on the bioaccumulation of kepone by sediment dwelling invertebrates are available. Adams (1987) reported sediment BAFs of 3 to 20 for chironomid midge larvae under flow-through conditions in sediments of 12.3 and 1.5 percent organic carbon, respectively. These two points allow the establishment of an estimated relationship between bioaccumulation and sediment organic carbon:

$$BAF_{inv} = -1.4\ (\%\ \text{organic carbon}) + 22.1$$

Applying this function to the 4 percent measured organic carbon content for Site-related sediments (i.e, Spring Creek) yields an estimated aquatic invertebrate sediment BAF of 16.5.

No data were available on the bioaccumulation potential of kepone in terrestrial invertebrates. The Connell and Markwell equation describing the bioaccumulation of a compound from soil/sediment to invertebrate (BAF_s) was applied as follows:

$$BAF_s = \frac{BAF_w}{K_p} = \frac{(\%\ \text{lipid})(K_{ow})^a}{(\%\ \text{Carbon})K_{ow}}$$

where: **a** is the non-linearity constant for bioaccumulation from water.

This fugacity equation yields a soil-to-earthworm BAF for kepone of 8.82 using the following values:

(1) K_{ow} 31,623 from log K_{ow} of 4.50;

(2) K_{oc} of 2,600;

(3) typical earthworm lipid content of 0.85 percent (Rao and Davidson 1980);

(4) assumed Site-area soil organic carbon of 5 percent (Manahan 1991); and

(5) a non-linearity constant (a) for organochlorine compound accumulation from water to earthworms of 1.14 (Lord et al. 1980).

d) Toxicity (Nonhuman)

Acute toxicity values (96-hour LC_{50}s) are reported for several fish species including channel catfish (0.225 mg/L), bluegill (0.072 mg/L), redear sunfish (0.044 mg/L), rainbow trout (0.030 mg/L), and unspecified trout (0.020 mg/L) (HSDB 1992). The toxic effects of kepone in fish suggest a neurological action characterized in the early stages by lethargy and loss of equilibrium (Schimmel and Wilson 1977). Severe toxicological effects can include continued equilibrium loss, sporadic hyperkinesis, scoliosis, development of black patches on the skin, tetanic convulsions and death (Couch et al. 1977; Roberts and Bendl 1982; Schimmel and Wilson 1977; Buckler et al. 1981). Kepone is also acutely toxic to daphnia (EC_{50} = 0.260 mg/L) in 7-day static bioassays. Toxic effects of kepone observed in aquatic invertebrates include reductions in growth, lethargic behavior, equilibrium loss, and death (USEPA 1978).

Little data on the chronic toxicity of kepone to freshwater organisms are available. Buckler et al. (1981) report a chronic no-observed-effect-

concentration (NOEC) of 0.31 μg/L for growth, survival, number of spawns, and egg production for the fathead minnow (*Pimephales promelas*). The authors also reported a chronic lowest-observed-effect-concentration (LOEC) of 3.1 μg/L for reduced fathead minnow fry survival. There are no reported studies that relate toxicity to aquatic species to levels of kepone in their food or in their own tissues as a result of bioaccumulation.

Some information is available on the toxicity of kepone in terrestrial invertebrates (i.e., honeybees and predaceous insects), but kepone is not particularly toxic to these organisms. Information is not available on kepone toxicity in amphibians and reptiles.

Kepone does not appear to be very acutely toxic to birds. This is evident in the LD_{50} values for mallard ducks (>2,400 mg/kg), bobwhite quail (530 mg/kg), ring-necked pheasant (600 mg/kg, adult; 115 mg/kg, young), and japanese quail (237 mg/kg) (HSDB 1992; USEPA 1978). However, there appears to be a potential for reproductive effects occurring in birds. Eroschenko and Wilson (1975) conducted a histological comparison study of reproductive organs, livers, and adrenal glands from immature and adult Japanese quail of both sexes fed 200 ppm kepone under two photoperiod regimes. Kepone exposure at this level had an estrogenic effect on the oviducts of immature females and on the testes of immature and mature males. Increased cellular proliferation, cytodifferentiation and tubular gland formation in the oviducts of exposed immature birds were observed. Kepone-treated birds exhibited hypertrophy of adrenal cortical and medullary cells. Exposed immature and adult male birds exhibited distended seminiferous tubules containing watery fluid.

Dewitt et al. (1962) reported reproductive effects in male ring-necked pheasants. The birds were exposed to kepone at doses of 50, 100, or 150 ppm in the diet. Birds at all dose groups exhibited the formation of female plumage, abnormal testes, malformed sperm, and elevated levels of reproductive failure. The LOEC for this study was determined to be 50 ppm kepone in the diet. No NOEC could be determined from this study

Naber and Ware (1965) reported that the addition of 75 or 100 ppm of kepone to the diets of female chickens resulted in significant reduction in egg production. The apparent hatchability of eggs was unaffected, but reduced embryonic survival was observed at a dietary concentration of 100 ppm

kepone. Survival of chicks 14 to 20 days post-hatch was also reduced, with 0 and 56 percent survival of chicks from hens fed 100 or 75 ppm kepone, respectively. The LOEC for this study has been determined to be 75 ppm kepone in the diet. No NOEC could be determined from this study.

McCall and Eroschenko (1988) exposed sexually mature Japanese quail to 40, 80, 160, or 200 ppm kepone in the diet for 60 days. Birds were sacrificed at 10, 30, and 60 days. There was a dose- and time-dependent reduction in germinal epithelium height in all birds following exposure to kepone. Exposure of quail to higher kepone doses (80, 160, and 200 ppm) for 30 or 60 days did not alter the ultrastructure of spermatogenic and Sertoli cells in the basal compartments. However, numerous maturing spermatid clumps located in the adluminal compartments of the germinal epithelium exhibited increased vacuolation, cytoplasmic degeneration, and desquamation at the three higher doses. The LOEC in this study is determined to be 80 ppm kepone in the diet, and the NOEC is determined to be 40 ppm.

Like other organochlorine pesticides, dietary kepone can cause teratogenic eggshell thinning. Defects were noted in the cuticle and vertical crystal layers of the eggshell of Japanese quail fed 225 ppm kepone in the diet for 21 days (USEPA 1975, as cited in USEPA 1978).

Symptoms of acute exposure in mammals include exaggerated startle responses, tremors, behavioral modifications, and weight loss (Guzelian 1982; Egle et al. 1979; Houston et al. 1981; Soine et al. 1982). Oral LD_{50} values of 95, 65, 250, and 2,550 mg/kg were observed in rats, rabbits, dogs, and pigs, respectively (HSDB 1992). Studies of the toxicological effects of chronic exposure to mammal species indicate the potential for neurological, hepatic, renal, and reproductive toxicity. Reported findings from repeat-dose studies are summarized in Appendix M.

D. Exposure Characterization

1. Sampling Results

Summaries of measured surface water, sediment, soil, and fish tissue concentrations of mirex and kepone are presented in Table 62.

a) **RMU1**

Surface soil samples collected from the Ruetgers-Nease property outside of the fenced area, specifically in the area where the spray field was previously located, indicated mirex concentrations of 32, 97, and 190 μg/kg for SS1, SS2, and SS3, respectively. Analysis of surface soil from SS1 and SS2 also identified kepone at 53 and 23 μg/kg, respectively. Kepone was not detected in SS3.

b) **RMU2A**

Surface water and/or sediments were collected from the drainage ditch which is further west and down-gradient from the west corner of the Ruetgers-Nease ground water treatment building (SW5, SW5-2/SW11 [a duplicate], SW7, SW10 and SED5, SED5-2/SED11 [a duplicate], SED7, SED10, and SS4). All of these sample locations are on Ruetgers-Nease property. Measured levels of mirex in surface water were: .045, .457/.500, not analyzed, and .483 μg/L, respectively, and in sediments: 6240, 905/1540, 1650, 1050, and 5.9 μg/kg, respectively. Measured levels of kepone in surface water were: ND (DL =.13), .061/.818, not analyzed, and ND (.13) μg/L, respectively, and in sediments: 667, ND/ND (68), ND (68), 58.6, and 51.7 μg/kg, respectively.

a) **RMU1**

Surface soil samples collected from the Ruetgers-Nease property outside of the fenced area, specifically in the area where the spray field was previously located, indicated mirex concentrations of 32, 97, and 190 μg/kg for SS1, SS2, and SS3, respectively. Analysis of surface soil from SS1 and SS2 also identified kepone at 53 and 23 μg/kg, respectively. Kepone was not detected in SS3.

TABLE 62
Mirex and Kepone Data Used in Ecological Risk Assessment
Ruetgers-Nease Chemical Site in State College, PA
(data are from the July 7, 1992 Draft RI Report except as noted)

	RMU1[a]		RMU2 (A and B)[b]		RMU3	
	Mirex	Kepone	Mirex	Kepone	Mirex	Kepone
SS1[c] µg/kg	32	53J				
SS2	97	23J				
SS3	190J	ND[1]				
Onsite:						
SW4 µg/l					.008J	.939
SW5			.045	ND[2]		
SW5-2/ SW11			.457/ .500	061J/ .818J		
SW10			.483	ND[2]		
Offsite:						
SW6			ND[3]	ND[2]		
SW8			.096	ND[2]		
Onsite:						
SED4 µg/kg					626	750J
SS4			5.9J	51.7J		
SED5			6240	667J		
SED5-2/SED11			905J/1540J	ND/ND[4]		
SED7			1650J	ND[4]		
SED10			1050	58.6J		
Offsite:						
SED6			185J	8J		
SED8			61.7J	ND[4]		
SED9			224	ND[4]		

TABLE 62
Mirex and Kepone Data Used in Ecological Risk Assessment
Ruetgers-Nease Chemical Site in State College, PA
(data are from the July 7, 1992 Draft RI Report except as noted)

	RMU4[a]		RMU5		RMU6	
	Mirex	Kepone	Mirex	Kepone	Mirex	Kepone
SW1[c] µg/l					ND[3]	ND[2]
SW2			ND[3]	ND[2]		
SW3	ND[3]	ND[2]				
SED1 µg/kg					36.9J	48.1J
SC-Benner[8]					26.9J	
SED2			42.4J	18.4J		
SC-Park[8]			73.4			
SED3	ND[5]	ND[4]				
SC-Background			ND[9]			
F1 µg/kg	110	330				
	ND[6]	ND[7]				
F2			330 170	550J ND[7]		
F3					180 110	500J ND[7]

NOTES:

[a] RMUs are Risk Management Units: RMU1 is the 15-acre mowed grassy field southwest of the Site; RMU2 is the intermittent flow drainage ditch; RMU3 is Thorton Spring; RMU4 is Spring Creek upstream of the Site; RMU5 is Spring Creek downstream of the Site in the Houserville Park area; and RMU6 is Spring Creek in the Benner Spring Fish hatchery area.
[b] A = on site, B = offsite
[c] SS = surface soil, SW = surface water, SED = sediment, and F = fish samples
[1] QL = 68
[2] QL = 0.13
[3] QL = 0.0054
[4] QL = 68
[5] QL = 18.5
[6] QL = 25
[7] QL = 190
[8] from Sept. 1992 sediment toxicity program; sample collected immediately downstream from Thornton Spring.
[9] DL = 6.5

b) RMU2A

Surface water and/or sediments were collected from the drainage ditch which is further west and down-gradient from the west corner of the Ruetgers-Nease ground water treatment building (SW5, SW5-2/SW11 [a duplicate], SW7, SW10 and SED5, SED5-2/SED11 [a duplicate], SED7, SED10, and SS4). All of these sample locations are on Ruetgers-Nease property. Measured levels of mirex in surface water were: .045, .457/.500, not analyzed, and .483 μg/L, respectively, and in sediments: 6240, 905/1540, 1650, 1050, and 5.9 μg/kg, respectively. Measured levels of kepone in surface water were: ND (DL =.13), .061/.818, not analyzed, and ND (.13) μg/L, respectively, and in sediments: 667, ND/ND (68), ND (68), 58.6, and 51.7 μg/kg, respectively.

c) RMU2B

Surface water and/or sediment was also collected from the drainage ditch downgradient from the Ruetgers-Nease property, along Route 26, and just prior to its confluence with Spring Creek (SW8 and SED8, SED9, and SW6 and SED6, respectively). Measured levels of mirex in surface water were: .096 and ND (DL =.0054) μg/L, respectively, and in sediments: 61.7, 224, and 185 μg/kg, respectively. Kepone was not detected in surface water samples; ND (.13) and ND (.13) μg/L, respectively. Kepone was detected in one of the three off-site drainage ditch sediments; ND (68), ND (68), and 8 μg/kg, respectively.

d) RMU3

Analysis of surface water collected from Thornton Spring immediately upstream from the culvert at Pike Street and prior to its confluence with Spring Creek (SW4) indicated concentrations of kepone (0.939 μg/l) and mirex (0.008 μg/l). Sediment collected from Thornton Spring (SED4) contained kepone (750 μg/kg) and mirex (626 μg/kg).

e) RMU4

Neither kepone nor mirex were detected in surface water (SW3) or sediment (SED3) collected from Spring Creek approximately 1,000 feet upstream from the confluence of Thornton Spring (SW3). Analysis of tissue

from a lower trophic level fish, the slimy sculpin (*Cottus cognatus*), from Spring Creek at the Pike Street Bridge in Lemont (F1L) indicated no detectable accumulation of either kepone or mirex. The analysis of tissue from a higher trophic level fish, the brown trout (*Salmo trutta*), from this upstream location (F1U), did however, indicate the presence of both kepone (330 μg/kg or ppb) and mirex (110 μg/kg or ppb).

f) RMU5

Kepone and mirex were not detected in surface water collected from Spring Creek in the area of Houserville Park (SW2). Sediment analyzed from this sampling location (SED2) in January 1991 indicated the presence of kepone (18.4 μg/kg) and mirex (42.4 μg/kg). A September 1992 sample collected in the same location indicated the presence of mirex (73.4 μg/kg), but no kepone was detected.

Analysis of slimy sculpin tissue from Spring Creek at Houserville Park (F2L) indicated concentrations of kepone (550 μg/kg) and mirex (330 μg/kg). Analyses of brown trout tissue from this location found no detectable kepone, but did find mirex at 170 μg/kg.

g) RMU6

Kepone and mirex were not detected in surface water collected from Spring Creek at the Benner Spring Fish Hatchery (SW1). Sediment collected from this location (SED1) in January 1991 indicated concentrations of kepone (48.1 μg/kg) and mirex (36.9 μg/kg). A September 1992 sample collected in the same location indicated the presence of mirex (26.9 μg/kg), but kepone was not detected.

Slimy sculpin (*Cottus cognatus*) fish tissue analyzed from this downstream location (F3L) indicated concentrations of kepone (500 μg/kg) and mirex (180 μg/kg). The analysis of brown trout (*Salmo trutta*) tissue (F3U) indicated no detectable kepone, but did indicate mirex (110 μg/kg).

2. Potential Exposure Pathways and Receptors

a) Water

Fish, macroinvertebrates, and aquatic plants inhabiting the drainage ditch, Thornton Spring and/or Spring Creek downstream of where the sources of chemical constituents enter the stream represent the primary potential receptors via direct exposure from the water compartment.

b) Sediment

As with the water compartment, fish, aquatic invertebrates, and rooted aquatic plants represent the primary potential receptors for direct exposure to chemicals present in sediment.

c) Foodchain

Because of the environmental stability and potential for bioconcentration and/or bioaccumulation associated with mirex and kepone, transfer of these chemicals in the foodchain is possible. Examples of dietary exposure to mirex or kepone through potential foodchain exposure routes are depicted below:

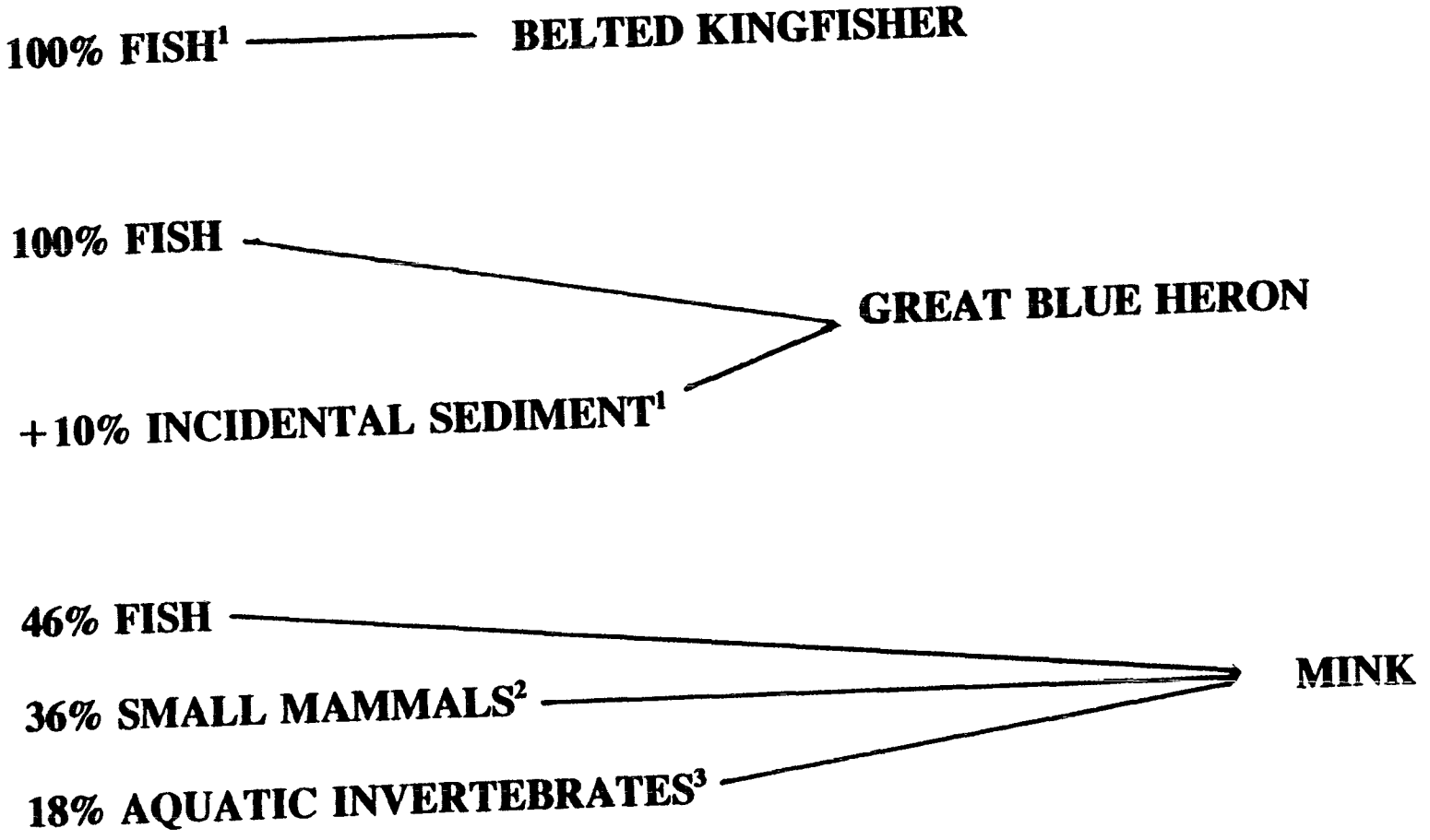
Aquatic foodchain: water and/or sediment → invertebrates and/or fish → piscivorous fish and/or wildlife

Terrestrial foodchain: soil → invertebrates and/or plants → omnivorous wildlife → wildlife predators

Exposure estimates were calculated for top-of-the-foodchain receptor organisms representative of each of the dietary exposure pathways outlined above for which there are applicable toxicity threshold data available (Figure 14). Exposures were not estimated for plant species because there are no available toxicity data for mirex and kepone (this is not considered a significant shortcoming in this assessment because "the plants" are the mowed grass field which is RMU1). Exposures were not estimated for aquatic

FIGURE 14

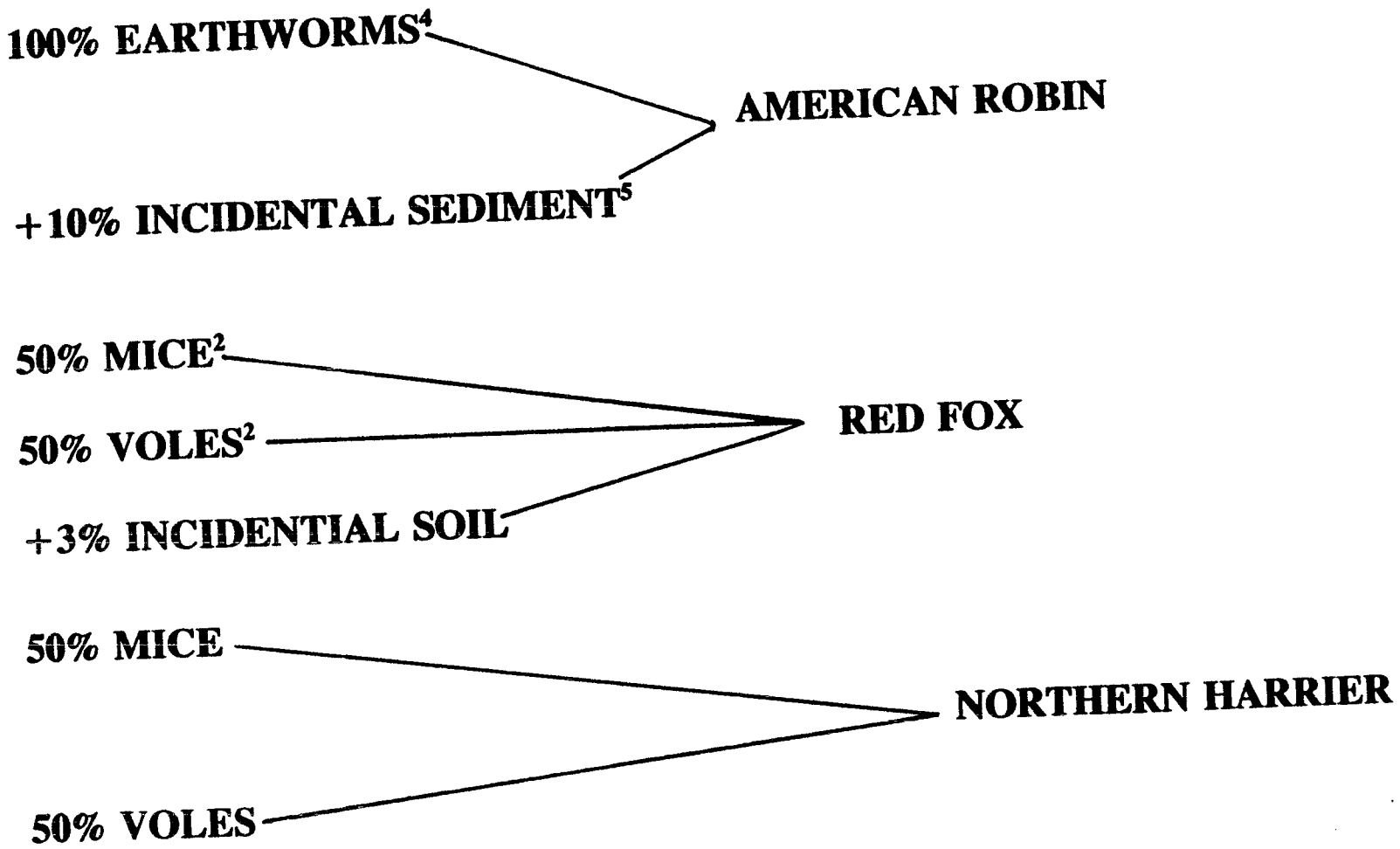
DIETARY EXPOSURE PATHWAYS FOR RECEPTOR SPECIES



[1] MEASURED RME CONCENTRATIONS
[2] ESTIMATE IN VOLES AND MICE FROM EATING WORMS (BAF FROM SOIL), VEGETATION (BAF FROM SOIL), AND INCIDENTAL SOIL (~25%) FROM AREAS WITH AS MUCH CHEMICAL AS IN RMU 1
[3] ESTIMATE FROM SEDIMENT BAF

FIGURE 14 (Continued)

DIETARY EXPOSURE PATHWAYS FOR RECEPTOR SPECIES



[4] ESTIMATE BASED ON BAF FROM RMU1 SOIL
[5] MEASURED RME CONCENTRATION

f:\alk\wp\statecol.epa

invertebrates or fish based on dietary intake of mirex or kepone because there are no toxicity data for these exposure pathways (e.g., levels in fish food or in fish tissues themselves that are related to specific toxicity endpoints). Exposures were not estimated for soil invertebrates or omnivorous wildlife because they do not represent the top of the foodchain where exposure is expected to be the highest.

d) Receptor Species - Selection

Receptor species are those species that are chosen to represent the larger biological community in the risk characterization step of the risk assessment. Selection criteria include: species which can reliably be considered to be present in the area (whether or not they have actually been observed); species that have a particular ecological, economic or aesthetic aspect in the area; and species that can, because of toxicological sensitivity or potential for varied types and magnitudes of exposure, be expected to represent the most sensitive populations in the area. There must also be toxicity data available that are directly applicable or can be reasonably extrapolated to the receptor species.

Receptor species for this assessment can be separated by RMUs and the habitats afforded by them.

- RMU1 (15-acre field) receptor species include: an insectivorous bird (the American robin), a raptor (the Northern harrier) and a carnivorous mammal (the red fox).

- RMU2 (intermittent flow drainage ditch) receptor species include: general aquatic species (e.g., benthic and/or water column macroinvertebrates primarily). No piscivorous species were selected as receptors as it is not likely that the ditch supports any fish or represents any other significant foodchain source. The on-site (RMU2A) and off-site (RMU2B) drainage ditch was treated the same with regard to receptor species.

- RMU3 (200-foot Thornton Spring) receptor species include sensitive aquatic species (e.g., macroinvertebrates). No piscivorous species were selected as receptors as the spring has a very limited foodchain source value given its small size and the close proximity of Spring Creek.

- RMU4 (upstream Spring Creek near Lemont) receptor species include: sensitive aquatic species; and the piscivorous Belted Kingfisher (100% fish diet),Great Blue Heron (fish and incidental sediment ingestion exposure), and the mink (fish, macroinvertebrate and small mammal diet).

- RMU5 (downstream of Thornton Spring and drainage ditch confluence) receptor species are the same as RMU4.

- RMU6 (further downstream near Benner Fish Hatchery) receptor species are the same as RMU4 and RMU5.

e) Receptor Species - Description

Great Blue Heron (*Ardea herodias*) - A piscivorous wading bird susceptible to chemical exposure from consumption of fish and incidental sediment ingestion. Although the blue heron is a consumer of fish, invertebrates, amphibians, reptiles, and small mammals (Martin et al. 1951), the assumption is made that fish is the only biological dietary component. This assumption allows the use of the measured residue data in fish and sediment which results in a maximum exposure estimate for this screening-level assessment.

Belted Kingfisher (*Ceryle alcyon*) - A diving bird which feeds primarily on fish, although invertebrates, reptiles, and amphibians have been observed in the diet (Martin et al. 1951). As with the heron, a 100 percent fish diet allows the use of measured tissue residue data which results in a maximum exposure estimate for this screening-level assessment.

Mink (*Mustela vison*) - A predator of both riparian and upland areas that feeds on fish, small mammals, invertebrates, and other animal matter (Chapman and Feldhamer 1982). Its exposure estimate for this screening-level assessment is based

on consumption of fish, aquatic invertebrates, and herbivorous and insectivorous small mammals. The fish tissue levels are those measured in the RMU being assessed; the other dietary components have an estimated level of mirex and kepone based on bioaccumulation models.

American Robin (*Turdus migratorius*) - A songbird that feeds upon terrestrial invertebrates such as the earthworm (Martin 1951). Although a significant portion (i.e., up to 80% seasonally) of the robin's diet is vegetative matter (Martin et al. 1951), the exposure estimate for this screening-level assessment is based on a 100 percent earthworm diet (earthworm levels of mirex and kepone based on bioaccumulation estimates from RMU1 soils) and on incidental ingestion of RMU1 soil.

Northern Harrier (*Circus cyaneus*) - A raptorial bird feeding in both wetland and upland areas. Craighead and Craighead (1969) indicate that this species feeds primarily on small mammals. Its exposure estimate for this screening-level assessment is based on a diet of herbivorous and insectivorous small mammal prey from RMU1 (plant and earthworm levels of mirex and kepone based on bioaccumulation estimates from RMU1 soils).

Red Fox (*Vulpes fulva*) - A terrestrial predator whose diet includes small mammals, terrestrial invertebrates, and plant material (Martin et al. 1951). Its exposure estimate is based on a diet of herbivorous and insectivorous small mammal prey from RMU1 (as for the harrier) and of incidental ingestion of soil from RMU1.

3. Exposure Calculations

a) Fish and Aquatic Invertebrates

Exposure levels for fish and aquatic invertebrates are the calculated reasonable maximum exposure levels (RMEs) based on measured levels of mirex and kepone in surface water and/or in sediments (Tables 63 and 64). RMEs are the 95th percent upper confidence limit of the mean, or the maximum, which ever is smaller (USEPA 1992). RMEs are based on samples collected within a particular RMU; data for more than one RMU were not combined. Measured levels for duplicate samples were averaged in the RME calculation.

TABLE 63
Surface Water Risks to Aquatic Organisms

RMU	Chemical	Surface Water RME (μg/1)	Surface Water Chronic Threshold[a] (μg/1)	Ratio RME/Threshold[b]
RMU2A	Mirex Kepone	.336 .19	0.001 0.1	336 1.9
RMU2B	Mirex Kepone	0.50 0.065	.001 0.1	50 0.65
RMU3	Mirex Kepone	0.0082 0.939	0.001 0.1	8.2 9.4
RMU4	Mirex Kepone	ND[c] ND	0.001 0.1	NA[d] NA
RMU5	Mirex Kepone	ND ND	0.001 0.1	NA NA
RMU6	Mirex Kepone	ND ND	0.001 0.1	NA NA

NOTES:

[a] Source mirex: Chronic AWQC
kepone: derived rom lowest chronic endpoint adjusted by appropriate uncertainty factors

[b] Ratios greater than 1 indicate potential risk

[c] Mirex and kepone were not detected in surface water in RMU4 through RMU6 (limit of detection for depone=0.132 μg/L; for mirex=0.0054 μg/L)

[d] NA = not applicable

TABLE 64
Sediment Risks to Benthic Organisms

RMU	Chemical	Sediment RME (µg/kg)	Sediment Threshold[a] (µg/kg)	Ratio RME/Threshold[b]
RMU2A	Mirex Kepone	2034 169	960 10.4	2.1 16.3
RMU2B	Mirex Kepone	157 25	960 10.4	0.16 2.4
RMU3	Mirex Kepone	626 750	960 10.4	0.7 72.1
RMU4	Mirex Kepone	ND[c] ND	960 10.4	NA[d] NA
RMU5	Mirex Kepone	73.4 18.4	960 10.4	0.08 1.8
RMU6	Mirex Kepone	36.9 48.1	960 10.4	0.04 4.6

NOTES:

[a] Source mirex: derived from the AWQC and equilibrium partitioning models with 4% sediment carbon
kepone: derived from the chronic aquatic effects threshold and equilibrium partitioning models with 4% sediment carbon

[b] ratios greater than 1 indicate potential risk

[c] ND = not detected

[d] NA = not applicable

TABLE 65
Foodchain Risks to Great Blue Heron

RMU	Chemical	Average Oral Exposure (μg/kg/day)	Bird Toxicity Threshold (μg/kg/day)	Ratio Exposure to Threshold[a]
RMU4	Mirex Kepone	22.0 66.1	3600 1000	<0.01 0.07
RMU5	Mirex Kepone	66.1 110	3600 1000	0.02 0.11
RMU6	Mirex Kepone	36.1 100	3600 1000	0.01 0.10
Average of all 3 RMUs	Mirex Kepone	41.5 92.1	3600 1000	0.01 0.09

NOTES:

[a] ratios greater than 1 indicate potential risk

TABLE 66
Foodchain Risks to Belted Kingfisher

RMU	Chemical	Average Oral Exposure (μg/kg/day)	Bird Toxicity Threshold (μg/kg/day)	Ratio of Exposure to Threshold[a]
RMU4	Mirex Kepone	55 165	3600 1000	0.01 0.17
RMU5	Mirex Kepone	165 275	3600 1000	0.05 0.28
RMU6	Mirex Kepone	90 250	3600 1000	0.03 0.25
Average of all 3 RMUs	Mirex Kepone	103 230	3600 1000	0.03 0.23

NOTES:

[a] Ratios greater than 1 indicate potential risk

TABLE 67
Foodchain Risks to Mink

RMU	Chemical	Average Oral Exposure (μg/kg/day)		Mammal Toxicity Threshold (μg/kg/day)	Ratio Exposure to Threshold[a]
		With RMU1[b]	Without RMU1[c]		
RMU4	Mirex Kepone	10.2 39.4	9.0 37.9	100 128	0.09-0.10 0.30-0.31
RMU5		35.1 47.7	33.9 46.2	100 128	0.34-0.35 0.36-0.37
RMU6		19.2 57.4	18.0 55.9	100 128	0.18-0.19 0.44-0.45
Average of all 3 RMUs		21.5 48.2	20.3 46.7	100 128	0.20-0.22 0.37-0.38

NOTES:

[a] ratios greater than 1 indicate potential risk
[b] mice and voles inhabiting RMU1 soils
[c] mice and voles inhabiting area with no mirex or kepone soil

TABLE 68
Foodchain Risks to American Robin

RMU	Chemical	Average Oral Exposure (μg/kg/day)	Bird Toxicity Threshold (μg/kg/day)	Ratio Exposure to Threshold[a]
RMU1	Mirex Kepone	12.8 52.7	3600 1000	<0.01 0.05

NOTES:

[a] Ratios greater than 1 indicate potential risk

TABLE 69
Foodchain Risks to Northern Harrier

RMU	Chemical	Average Oral Exposure (μg/kg/day)	Bird Toxicity Threshold (μg/kg/day)	Ratio Exposure to Threshold[a]
RMU1	Mirex Kepone	0.06 0.1	3600 1000	<0.01 <0.01

NOTES:

[a] Ratios greater than 1 indicate potential risk

TABLE 70
Foodchain Risks to Red Fox

RMU	Chemical	Average Oral Exposure (μg/kg/day)	Mammal Toxicity Threshold (μg/kg/day)	Ratio Exposure to Threshold[a]
RMU1	Mirex Kepone	<0.1 1.6	100 128	<0.01 0.01

NOTES:

[a] Ratios greater than 1 indicate potential risk

b) Birds and Mammals

The exposure models and calculations of mirex and kepone exposures for the 6 receptor species are presented in Appendix N. The mean measured media concentrations of mirex or kepone in fish, sediment, or soil were incorporated into the exposure estimates as indicated. These same mean measured levels were used to estimate tissue levels in earthworms, plants, and herbivorous and insectivorous small mammals as warranted. Tables 65-70 present the estimated daily dose of mirex or kepone to each receptor species for each of the RMUs. These exposure estimates are likely to be overestimates of actual daily exposure. Conservative assumptions include: 1) the use of maximum proportions of dietary components that would contain "high levels" of mirex and kepone (e.g., the 100% fish diet of the heron and kingfisher); and 2) that all of the diet comes from that area with the RME level of mirex or kepone. A number of the receptor species have foraging ranges and seasonal behaviors that would very likely take them beyond those areas that have measurable levels of mirex or kepone (because RMU1 is relatively small, only 15 acres, the actual foraging ranges of the red fox and northern harrier were used to estimate dietary exposures). The conservative assumptions and resulting exposure estimates are intended as a screening-level evaluation which could be refined to be more reflective of the actual site situation if warranted.

E. Toxicity Thresholds for Mirex and Kepone

In addition to the ecological toxicology data presented in Section D, there are several published reviews which systematically evaluate the available data and offer conclusions regarding toxicity thresholds (or safe levels) for mirex and kepone. Thresholds have been estimated for exposures from water, sediments, and wildlife foodchains.

1. Mirex

a) Water

USEPA (1986c) has established a chronic ambient water quality criterion (AWQC) of 0.000001 mg/L (one part per trillion) mirex for freshwater aquatic life. This criterion is based upon an uncertainty factor of 0.01 (or 1%) applied to the lowest level at which effects were observed on

sensitive invertebrates including crayfish. The effect was delayed mortality seen at 0.0001 mg/L mirex. The uncertainty factor is intended to account for serious sublethal effects and the possibility of more sensitive species than those tested.

b) Sediment

USEPA (1988c) has proposed a procedure, known as the "equilibrium partitioning procedure," for using the chronic AWQC and the K_{oc} value of a chemical to derive an interim sediment quality criterion for the protection of aquatic life. These criteria, which are estimated chronic toxicity thresholds, have been derived by USEPA for only a few organic chemicals; mirex is not one of them. However, the agency's approach, which was developed especially for non-polar hydrophobic organics, can be applied to mirex to provide an estimate of a chronic toxicity threshold.

The equilibrium partitioning procedure is based upon the assumption that the sediment pore water concentration of a chemical is the bioavailable and potentially toxic fraction for benthic organisms. The sediment pore water concentration is dependent upon the partitioning of the chemical between water and sediment organic carbon. The formula for the application of the approach is as follows:

$$\text{Sediment Quality Criterion} = (\text{AWQC } \mu g/L)(K_{oc}) \,/\, 1{,}000 g/kg \text{ in } \mu g/g \text{ carbon}$$

This formula yields an estimated chronic sediment toxicity threshold of 24 μg/g carbon for mirex using the 0.001 μg/L AWQC and a K_{oc} value of 24,000,000. The 24 μg/g value is then adjusted to account for the organic carbon concentration in the sediments under study. For example, at 4 percent organic carbon, the toxicity threshold value becomes 960 μg/kg (or 0.96 ppm mirex) via 24 μg mirex/gC X 40 gC/kg sediment.

These "interim sediment quality criteria" values are only estimates of chronic toxicity thresholds in sediments; and, because the approach is based on USEPA's acknowledged conservative assumptions, it should be used as a screening tool rather than as providing definitive criteria for sediment remediation or other regulatory purposes. If the measured (or predicted)

sediment levels exceed the estimated sediment toxicity threshold, then additional refinement of the exposure and/or toxicity values may be appropriate. Alternatively, sediment toxicity investigations may provide a better evaluation of chemical bioavailability and potential risk at the site.

c) Edible Tissues for Wildlife Foodchains

For the purposes of this assessment, mirex chronic toxicity thresholds have been developed for wildlife (birds and mammals) that may be exposed via consumption of food, soils, and/or sediments containing mirex. The edible tissue and dietary intake thresholds have been developed using the conservative assumption that 100 percent of the animal's diet will contain mirex. This assumption can be modified using Site-specific data as needed.

The threshold estimates should be considered as a screening tool rather than as definitive criteria for site remediation or other regulatory purposes. Exceedence of an edible tissue or dietary threshold would suggest that the potential exists for toxic effects in wildlife. Actual occurrence of effects would depend on the extent to which individual animals consume food with residues exceeding the predicted toxicity threshold and the duration of consumption/exposure.

1) Uncertainty Factors

The thresholds were developed using an uncertainty factor (UF) scheme developed by Newell et al. (1987) of the New York State Department of Environmental Conservation for the estimation of fish flesh criteria for piscivorous wildlife. These UFs are applied to the no-observed-adverse-effect levels (NOAELs) or to the lowest-observed-adverse-effect levels (LOAELs) reported in toxicity tests to account for differences in interspecies sensitivities to a given chemical and for limitations in the available experimental data. The Newell et al. UF scheme is as follows:

Uncertainty Factors (UFs)

Interspecies Adjustment -

- If 3 or more species' NOAELs in a class exist, UF = 1

(i.e., the lowest NOAEL can serve as the wildlife estimate NOAEL)
- If 1 or 2 species' NOAELs in a class exist, UF = 0.1

Short-term Versus Long-term Adjustment -
- For chronic studies (i.e., >90 days exposure), UF = 1
- For shorter-term studies (i.e., 30-90 days), UF = 0.1

LOAEL to NOAEL Adjustment -
- For reported NOAELs, UF = 1
- To estimate a NOAEL from a LOAEL, UF = 0.2

2) Bird Chronic Toxicity Threshold

A review of available subchronic and chronic mirex toxicity data for birds is presented in Section D.1.d and Appendix K. The edible tissue chronic toxicity threshold for birds is estimated to be 40 mg mirex/kg food (ppm) in the diet. This toxicity threshold expressed on a daily intake basis is 3.6 mg mirex/kg body weight/day.

The edible tissue threshold is based on the study by Kendall et al. (1978). Kendall et al. exposed bobwhite quail to mirex in feed at concentrations of 0, 20, or 40 mg/kg (ppm) beginning at 1 day of age through the grow-out and egg-laying periods. At least 5 breeding pairs per dose group (F_0 generation) were carried through breeding. At 4 bi-weekly intervals, eggs were incubated. Hatchability and chick survival were determined. The F_0 generation was sacrificed at 36 weeks, and one replicate of F_1 hatchlings receiving dietary mirex was grown out for investigation of reproductive potential of second generation treatment birds. Because of predation in the F_1 generation, only the 1 mg/kg F_1 group was carried through breeding. The investigators reported no treatment-related effects on productivity, on survival of quail embryos to 3 weeks, egg hatchability, chick survival to 2 weeks, or survival of the F1 generation through grow-out and egg laying. Again, only the 1 mg/kg F_1 generation group was available for the egg production phase, and no problems with embryonation, embryo survival, and hatchability were detected.

Given the absence of adverse effects in any of the treatment groups, the highest dose level tested in the Kendall et al. (1978) study, 40 mg/kg in the diet, is considered to be the NOAEL. This NOAEL is consistent with the findings of studies in other bird species. Hyde (1972) reported effects on duckling survival at 100 ppm mirex in the diet, but not at 1 ppm. In other studies of mirex in birds, effects have been observed only at dietary concentrations above 100 ppm.

An edible tissue threshold for birds was derived from the NOAEL of 40 mg/kg (ppm) using the UF scheme of Newell et al. (1987). Because at least three species NOAELs in birds were reported in the literature, and because the Kendall et al. (1987) study was of chronic duration (i.e, >90 days) and identified a NOAEL, no adjustments for uncertainty need to be applied. The edible tissue threshold therefore remains 40 mg/kg (ppm) in the diet of birds.

For purposes of the risk assessment, a threshold value in units of mg mirex per kg bird body weight per day (mg/kg/day) was also calculated. According to USEPA (1986) (HED, Standard Evaluation Procedure, Ecological Risk Assessment), the body weight and food consumption rate for bobwhite quail is 0.17 kg and 0.0152 kg/day, respectively. The toxicity threshold can therefore be calculated as:

$$\frac{40 \text{ mg/kg diet} \times 0.0152 \text{ kg diet/day}}{0.17 \text{ kg}} = 3.6 \text{ mg/kg/day}$$

3) Mammal Chronic Toxicity Threshold

A review of available subchronic and chronic mirex toxicity data for mammals is presented in Section D.1.d and Appendix L. Based on this review, the reproductive toxicity study of Chu et al. (1981) was selected as the basis for the chronic toxicity threshold. The edible tissue threshold for mammals is estimated to be 1 mg mirex/kg food in the diet. This toxicity threshold expressed on a daily intake basis is 0.1 mg/kg/day for a small mammal.

Chu et al. (1981) fed groups of rats diets containing 0, 5, 10, 20, or 40 mg/kg (ppm) mirex for 13 weeks prior to mating, during a 2-week mating period, and through gestation and lactation. Toxicity in

adult females was assessed based on weight gain, hematologic analyses, serum chemistry, liver enzymes, and histopathology. Pups were evaluated with respect to body weight (at birth, 4, and 21 days), survival, and histopathology at 21 days. Adult females at 40 mg/kg showed a significant decrease in weight gain. Litter size was significantly decreased in all treatment groups, and 21-day pup survival was affected at 20 mg/kg. Enlarged livers were observed in adults at 40 mg/kg. Histopathological changes in the livers and thyroids of mothers and pups were observed in all treatment groups, and cataract formation was reported in pups at 5 mg/kg. The LOAEL in this study is therefore the lowest dose (5 mg/kg); a NOAEL was not identified.

The Chu et al. (1981) study was selected as the basis for the toxicity threshold because reproductive endpoints are particularly relevant to an ecological risk assessment. Other chronic toxicity studies (summarized in Appendix L), which generally evaluated histopathologic changes associated with mirex exposure, suggest that the lowest reported NOAELs are about 1 to 2 mg/kg in the diet, consistent with the LOAEL in the Chu et al. study of 5 mg/kg.

A chronic toxicity threshold for mammalian wildlife in units of mg/kg/day was derived from the LOAEL of 5 mg/kg reported in Chu et al. (1981) using dose conversions (i.e., mg/kg diet to mg/kg body weight/day) provided in IRIS (1992) and the UF scheme based on Newell et al. (1987). According to IRIS (1992), the dietary concentration of 5 mg/kg reported by Chu et al (1981) is equivalent to a dose of 0.5 mg/kg/day. An interspecies adjustment was not needed because there are mirex studies for at least 3 species (rats, mice, and dogs) reported in the literature, and the most sensitive species was used in the assessment. Because the lowest test dose in Chu et al. (1981) was a LOAEL, a UF of 0.2 was applied. The resulting chronic toxicity threshold is:

$$0.5 \text{ mg/kg/day} \times 0.2 = 0.1 \text{ mg/kg/day}$$

2. Kepone

a) Water

USEPA has not established a water quality criterion for kepone in either freshwater or marine waters. Consequently, a threshold toxicity value for kepone in freshwater has been estimated on the basis of the available aquatic toxicity data presented in Section D of this chapter.

Buckler et al. (1981) report a chronic NOEC and LOEC for survival of fathead minnow fry (*Pimephales promelas*) of 0.31 ppb and 3.1 ppb, respectively. A maximum acceptable toxicant concentration (MATC) for this study can be derived by taking the geometric mean of the LOEC and the NOEC, resulting in a MATC of 0.98 ppb.

Holcomb et al. (1988) regressed toxicity data for 44 aquatic species (fish, amphibians, and invertebrates) against toxic responses in fathead minnows and found that the responses of most aquatic species fell within one order of magnitude from the fathead minnow response. Therefore, an aquatic chronic toxicity threshold for kepone in aquatic organisms can be calculated by dividing the fathead minnow MATC of 0.98 ppb by an interspecies extrapolation factor of 10 to yield a chronic aquatic toxic threshold of 0.1 ppb.

b) Sediment

The same USEPA procedure (i.e., equilibrium partitioning; USEPA 1988c) used to estimate the equivalent of an interim sediment quality criteria for mirex (see Section F.1.b) was used for kepone.

USEPA (1988c) has proposed a procedure, known as the "equilibrium partitioning procedure," for using the chronic AWQC and the K_{oc} value of a chemical to derive an interim sediment quality criterion for the protection of aquatic life. These criteria, which are estimated chronic toxicity thresholds, have been derived by USEPA for only a few organic chemicals; kepone is not one of them. However, the agency's approach, which was developed especially for non-polar hydrophobic organics, can be applied to kepone to provide an estimate of a chronic toxicity threshold.

The equilibrium partitioning procedure is based upon the assumption that the sediment pore water concentration of a chemical is the bioavailable and potentially toxic fraction for benthic organisms. The sediment pore water

concentration is dependent upon the partitioning of the chemical between water and sediment organic carbon. The formula for the application of the approach is as follows:

Sediment quality criterion = (aquatic tox. threshold μg/L)(K_{oc}) / 1,000 g/kg in μg/gC

Using the estimated aquatic chronic toxicity threshold of 0.1 μg/L and the HSDB (1992) K_{oc} estimate of 2,600 yields an estimated chronic sediment threshold of 0.26 μg/gC for kepone. This value is then adjusted to account for the organic carbon concentration in the sediments under study. For example, at 4 percent organic carbon, the toxicity threshold value becomes 10.4 μg/kg (via 0.26 μg kepone/gC X 40 gC/kg sediment).

These "interim sediment quality criteria" values are only estimates of chronic toxicity thresholds in sediments; and, because the approach is based on USEPA's acknowledged conservative assumptions, it should be used as a screening tool rather than as providing definitive criteria for sediment remediation or other regulatory purposes. If the measured (or predicted) sediment levels exceed the estimated sediment toxicity threshold, then additional refinement of the exposure and/or toxicity values may be appropriate. Alternatively, sediment toxicity investigations may provide a better evaluation of chemical bioavailability and potential risk at the site.

c) Edible Tissues for Wildlife Foodchains

For the purposes of this assessment, kepone chronic toxicity thresholds have been developed for wildlife (birds and mammals) that may be exposed via consumption of food, soils, and/or sediments containing kepone. The edible tissue and dietary intake thresholds have been developed using the conservative assumption that 100 percent of the animal's diet will contain kepone. This assumption can be modified using Site-specific data as needed.

The threshold estimates should be considered as a screening tool rather than as definitive criteria for site remediation or other regulatory purposes. Exceedance of an edible tissue or dietary threshold would suggest that the potential exists for toxic effects in wildlife. Actual occurrence of effects would depend on the extent to which individual animals consume food with

residues exceeding the predicted toxicity threshold and the duration of consumption/exposure.

1) Uncertainty Factors

The thresholds were developed using an UF scheme developed by Newell et al. (1987) of the New York State Department of Environmental Conservation for the estimation of fish flesh criteria for piscivorous wildlife. These UFs are applied to the NOAELs or to the LOAELs reported in toxicity tests to account for differences in interspecies sensitivity to a given chemical and for limitations in the available experimental data. The Newell et al. UF scheme is as follows:

Uncertainty Factors (UFs)

Interspecies Adjustment -

- If 3 or more species' NOAELs in a class exist, UF = 1
  (i.e., the lowest NOAEL can serve as the wildlife estimate NOAEL)
- If 1 or 2 species' NOAELs in a class exist, UF = 0.1

Short-term Versus Long-term Adjustment -

- For chronic studies (i.e., >90 days exposure), UF = 1
- For shorter-term studies (i.e., 30-90 days), UF = 0.1

LOAEL to NOAEL Adjustment -

- For reported NOAELs, UF = 1
- To estimate a NOAEL from a LOAEL, UF = 0.2

2) Bird Chronic Toxicity Threshold

A review of available subchronic and chronic kepone toxicity data for birds is presented in Section D.2.d. The edible tissue chronic toxicity threshold for birds is estimated to be 40 mg kepone/kg food (ppm) in the diet. This toxicity threshold expressed on a daily intake basis is 1 mg kepone/kg body weight/day.

The edible tissue threshold is based on the study by DeWitt et al. (1962). DeWitt et al. exposed ring-necked pheasants to kepone in feed at concentrations of 0, 50, 100, or 150 mg/kg (ppm). Birds exposed to kepone exhibited development of adult female plumage accompanied by abnormal testes, malformed sperm, and reproductive failure. The LOAEL in this study was determined to be 50 ppm.

An edible tissue threshold for birds was derived from the LOAEL of 50 mg/kg (ppm) using the UF scheme of Newell et al. (1987). Because at least 3 species effect levels in birds were reported in the literature, and because the Dewitt et al. study was of chronic duration (i.e., >90 days) and identified a LOAEL, only an adjustment for LOAEL to NOAEL (0.2) was applied. The edible tissue threshold is therefore 10 mg/kg (ppm) in the diet of birds.

For purposes of the risk assessment, a threshold value in units of mg kepone per kg bird body weight per day (mg/kg/day) was also calculated. According to personal communication with Curt Hutchinson (Wildlife International), the maximum weight of tested ring-necked pheasants is 1.5 kg and the daily dietary consumption rate is 10 percent of the body weight or approximately 0.150 kg. The toxicity threshold can therefore be calculated as:

$$\frac{10 \text{ mg/kg diet} \times 0.150 \text{ kg diet/day}}{1.5 \text{ kg}} = 1.0 \text{ mg/kg/day}$$

3) Mammal Chronic Toxicity Threshold

A review of available subchronic and chronic toxicity data for mammals is presented in Section D.2.d and Appendix M. Based on this review, the reproductive toxicity study of Good et al. (1965) was selected as the basis for the chronic toxicity threshold. The edible tissue threshold for mammals is estimated to be 0.1 mg kepone/kg diet. This toxicity threshold expressed on a daily intake basis is 0.128 mg/kg/day.

The Good et al. (1965) study was selected as the basis for the toxicity threshold because reproductive endpoints are particularly relevant to an ecological risk assessment. Good et al. (1965) found a

statistically significant reduction in the reproductive success of mice fed a diet containing kepone at a concentration of 5 ppm (0.64 mg/kg/day). The progeny of these exposed mice (F_1), when bred, also produced significantly lower litters than controls. Decreased litter production in F_1 mice occurred regardless of whether they were exposed to kepone at 5 ppm or not. The LOAEL in this study is therefore the lowest dose (5 mg/kg) and can also be expressed in terms of a daily oral dose (0.64 mg/kg/day).

A chronic toxicity threshold for mammalian wildlife was derived from the LOAEL of 5 mg/kg (0.64 mg/kg/day) using the uncertainty factor scheme of Newell et al. (1987). An interspecies adjustment was not needed because there are kepone studies for at least three species reported in the literature, and the most sensitive species was used in the assessment. Because the lowest test dose in Good et al. (1965) was a LOAEL, a UF of 0.2 was applied. The resulting chronic toxicity threshold is:

5 mg/kg diet X 0.2 = 0.1 mg/kg diet, or

0.64 mg/kg/day X 0.2 = 0.128 mg/kg/day

F. Risk Characterization

There are numerous data available that can be used to characterize the potential ecological risks associated with the Site. These include: the RI chemical monitoring data in soil, water, sediment, and fish; the historical sediment and fish tissue monitoring data; the water and sediment bioassay data; the historical fish population and benthic community data; and finally, the "ecotoxicological data" (i.e, chemical-specific toxicity thresholds, and fate and exposure-modeling data based on the published literature). In this assessment, the ecotoxicological data are used to estimate potential risks based on the hazard quotient method (Rodier and Mauriello, 1993). This generally accepted screening level approach for toxic chemicals compares estimates of toxicity to RI monitoring-based exposure for each chemical of concern in order to determine if exceedances (and therefore potential risks) are predicted for a representative group of indicator or receptor species. The outcome of this analysis is then combined with the site-specific data on bioassays and the historic and present condition

of the biological community of the study area to address the likelihood that adverse effects have occurred or are occuring.

The risk characterization is done on the basis of the RMUs (logical subunits of the Site based on environmental media and proximity to sources of the chemicals of concern). The hazard quotients are calculated for each receptor in each RMU. A quotient ratio of less than one (i.e., exposure is less than the toxicity threshold) suggests negligible risk, and a quotient of greater than one suggests that a risk is possible. Using the quotient method, priority and resources can be applied to those scenarios where there is the highest likelihood of risk. The assessment, even at this screening level, does attempt to address 1) what exposure pathways exist; 2) what receptor populations could be at risk; and 3) what is the relative magnitude of the exposure and the severity and time frame of the potential ecological effects.

The estimates of environmental exposures for the aquatic and terrestrial biota of the RMUs are based on RME values. The RMEs represent the upper 95th percentile confidence limit of the mean chemical concentration in the medium of interest, or the maximum measured concentration, if it is below the calculated 95th percentile (see Chapter VI). It is reasonable to use the RME analysis as a "worst case" estimate for the ecological assessment, considering that the potential receptor species are mobile and likely to range beyond a single sampling station, and that the toxicity threshold values are related to long-term, chronic exposures.

The RME values for mirex and kepone (by RMU and media) are presented in Tables 63-70. For small mammals, songbirds, and predators in the terrestrial foodchain, exposures are estimated from measured soil levels. For aquatic species and fish-eating wildlife, RMEs are based on measurements or on estimates in the water, sediments, and/or fish tissues.

The ecotoxicology data for mirex and kepone are reviewed in Section C. The chronic toxicity thresholds for each of the receptor species, whether an ARAR or an estimate using available data and methods, are presented in Section E.

The results of quotient analyses, by receptor species and RMU, are presented in Tables 63-70 for mirex and kepone. These results combined with the available site-specific information (i.e, bioassays and historical and present conditions) are used to develop the following conclusions about the potential environmental risks of mirex and kepone (and any other chemicals not addressed individually) at the Site:

1. RMU1 (the 15-acre mowed-grass field)

There are negligible risks to terrestrial wildlife (e.g., American robin, northern harrier, red fox, and mink) as determined by maximum exposure estimates from on-site soils. The RME exposure values used in the risk estimates were 190 μg/kg (ppb) mirex and 53 μg/kg kepone. These values were also the highest surface soil levels in the former spray field area; both were "J" qualified estimates. The exposure models included foodchain uptake through earthworms, vegetation, mice, and voles, and via incidental soil ingestion. The quotient ratios ranged from <0.01 to 0.05.

The percent of diet containing mirex or kepone was adjusted for the northern harrier and red fox to account for their foraging ranges (see Appendix N). The robin diet was made up of 100 percent earthworms from RMU1.

Plant species were not factored into the assessment because there are no available phototoxicity data for mirex and kepone. The predominant plants in RMU1 are grasses which, based on direct observation, appear not to be stressed or otherwise impacted by these chemicals in the soil.

2. RMU2 (the drainage ditch)

a) RMU2A - drainage ditch on site

The quotient method predicts the potential for some level of chronic toxicity risk to sensitive aquatic organisms exposed to mirex in the water column and to mirex and kepone in the sediments of RMU2A (the on-site drainage ditch). Predicted exposures exceeded toxicity thresholds by 336 (mirex) and 1.9 (kepone) for water column organisms, and by 2.1 (mirex) and 8.1 (kepone) for sediment dwelling organisms. It is possible, however, that the approach used overestimates the potential for adverse ecological effects in the ditch. This is because the ambient water quality criteria and estimated "sediment criteria" which are used in the assessment are derived for the protection of very sensitive aquatic species that may not be representative of the aquatic species inhabiting the drainage ditch. As discussed in Section 4b, the intermittent flow of water in the ditch results in the use of this area by terrestrial (e.g., earthworms) and semi-aquatic (e.g., snails) species rather than strictly aquatic species (e.g., fish). Therefore, it is possible that the assessment overestimates risks. In the case of relatively low exceedances (i.e.,

1.9, 2.1 and even 8.1), it is likely that the risk to those species inhabiting the drainage ditch is very low. The exceedence of 336 for mirex in the water is relatively high, however, and indicates potential risk for species inhabiting the ditch.

Ground water treatment facility effluent which empties into the drainage ditch on-site was tested for toxicity to the sensitive fathead minnow and Ceriodaphnia invertebrate in a bioassay. The sample was not toxic to the invertebrate; the growth of the fish was reduced by the effluent, but the toxicity was eliminated when the effluent was diluted by fifty percent. Because fish do not (and, under all but very high water conditions, could not) inhabit the drainage ditch on-site, and because the invertebrate species that was tested was not affected, the bioassay results support the conclusion above that the predictions of risk from the quotient method very likely overestimate the actual risk from mirex and kepone to biota in the drainage ditch on site. Because the effluent bioassay would test for toxicity to all the chemicals contained therein, the conclusion of minimal risk carries over to the VOCs in the drainage ditch water.

b) RMU2B - the drainage ditch off site

The quotient method predicts negligible risk for mirex in sediment (ratio of 0.16) and for kepone in water (0.65), marginal risks for kepone in sediment (2.4), and higher risks for mirex in water (50.0) in the off-site portion of the drainage ditch. As was presented above for the on-site ditch, however, those risks may represent overestimates given that the terrestrial and semi-aquatic species that inhabit the ditch might not be as sensitive as the aquatic species upon which the toxicity thresholds were based.

A drainage ditch sample collected above the confluence of Spring Creek was used in a bioassay with the fathead minnow and Ceriodaphnia invertebrate. There was no toxicity observed for the invertebrate. The survival and growth of the fish was reduced by the ditch water, but the toxicity was eliminated when the water was diluted. The results suggest that fish might be at risk if they inhabit the ditch in its lower reaches, but that the toxicity should not carry over into Spring Creek because of dilution. Again, this bioassay would cover VOCs or any other chemicals that were present at the time the samples were collected.

Because the drainage ditch habitat (both on-site and off-site) is not likely to support a significant aquatic biota, there is negligible risk for piscivorous wildlife that would consume aquatic organisms from the ditch. Wildlife that feed on soil invertebrates (e.g., birds eating earthworms, or a raccoon scavenging for food) could however be exposed to some mirex and/or kepone coming from ditch sediments/soils. While a specific exposure scenario was not developed for wildlife that would feed in the ditch, the results of the RMU1 evaluation are applicable. The estimated exposues for wildlife that might derive all their food from RMU1, the 15-acre field, were 20 to >100 fold below chronic toxicity thresholds. Negligible risks were predicted. While the levels of mirex and kepone in the on-site drainage ditch sediments are up to ten times higher than the levels in RMU1 surface soils, the potentially higher exposure for wildlife feeding in the ditch would still result in quotient ratios of less than one. The quotient ratios for wildlife in the off-site drainage ditch area would be even less than for the on-site drainage ditch area because of lower levels of mirex and kepone. Therefore, no significant risks from mirex or kepone are predicted for wildlife inhabiting the area of the drainage ditch.

3. RMU3 (Thornton Spring)

The quotient method predicts the potential for toxicity risks to sensitive aquatic organisms associated with mirex and kepone in Thornton Spring. These risks are, in practical terms, limited to invertebrates because Thornton Spring is unlikely to provide a habitat that would support a fish population. Thornton Spring discharges into Spring Creek via a culvert pipe that is above the normal water level of the creek; this would keep fish from entering the spring. At RME levels, the quotients for water column exposures are approximately 8 and 9 for mirex and kepone, respectively. Mirex is not predicted to result in a risk based on sediment exposures. Kepone in sediment had a quotient ratio of 72 which infers some risk, however, the RME exposure level used in the risk calculation was based on a "J" qualified analysis and therefore the 72 value is not definitive.

The result of both the Thornton Spring surface water bioassays provide additional perspective to the results of the quotient analysis. At 100 percent concentration, Thornton Spring water was observed to be toxic (non-lethal effects) to sensitive fish and invertebrate species. This is consistent with the exceedences in

water predicted for mirex and kepone using the quotient approach. In addition, toxicity of the spring water would also reflect any toxicity from VOCs if they were present. However, toxicity was eliminated once the water samples were diluted, and because Thornton Spring represents about one percent of the flow of Spring Creek, the prediction of surface water toxicity would not carry over into Spring Creek.

Even more directly relevant to the assessment of risk are the biological observations made of Thornton Spring during the RI sampling. Invertebrates (including the sensitive crayfish, plus earthworms and midges) were found inhabiting the spring. This implies that the toxicity results from the laboratory bioassays may not translate directly to the conditions in Thornton Spring; toxicity of the water and the sediment does not appear to prevent sensitive aquatic species from living there. That is, the risks may be overpredicted using either the quotient method or the bioassays.

A risk characterization for piscivorous wildlife was not considered to be applicable for Thornton Spring. The spring is only approximately 200 feet in length; its highly variable flow and confluence through a culvert pipe above Spring Creek precludes it from being a productive aquatic habitat, at least in that it would not support piscivorous wildlife. A predator may occasionally look for food in Thornton Spring, but there is much more productive (and presumably more attractive) habitat in adjacent Spring Creek.

4. RMU4 (upstream of the Site on Spring Creek)

Both mirex and kepone were below detection limits in surface water and sediment samples (historically, the sediments have been below detection - Figure 3) in the RMU4 (Lemont) area. Aquatic organisms are not predicted to be at risk from mirex or kepone in the water or in sediments. The surface water bioassays with fish and Ceriodaphnia, and the sediment toxicity tests with amphipods and midge (i.e., SC-Background sample) generally showed no toxicity, thus, supporting the conclusion of negligible risks. The only exception was toxicity observed in one of the two fish tests with surface water; it is unknown if this result reflects variability in the test or a possible upstream source of chemicals at toxic levels. The available historical data on fish and benthic macroinvertebrates (Figures 11-13) indicate variable but healthy populations in the Lemont area upstream of the Site.

Part per billion concentrations of mirex and kepone were detected in upper trophic level fish collected in the RMU4 area. It is difficult to interpret the

ecotoxicological significance of fish tissue concentrations because data on the relationship of body burden to toxic effects are lacking for mirex and kepone. Beyond that for wildlife (discussed in the next paragraph), the only tissue criteria that are available are the FDA action levels. The FDA action levels, 0.1 ppm mirex and 0.3 ppm kepone, are established to protect humans. The fish tissue levels in the RMU4 area were 0.11 ppm mirex and 0.33 ppm kepone, both being at the FDA action levels. While there are no historical data showing trends for fish tissue levels in the RMU4 area, the general trend has been downward at all sites on Spring Creek that have been monitored (Figure 10).

The quotient analysis predicts negligible risks to piscivorous wildlife (e.g., heron, kingfisher and mink) from exposure via a diet of 100 percent fish and invertebrates at the RME levels of mirex or kepone plus the added exposure of incidental sediment ingestion for the heron. The quotients range from < 0.01 to 0.31 under maximum exposure conditions. RMU4 had the lowest levels of mirex and kepone in fish for any of the Spring Creek sampling locations.

5. RMU5 (Spring Creek below Thornton Spring and drainage ditch)

a) Surface Water

Both mirex and kepone were below detection limits (0.0054 ug/L and 0.132 ug/L, respectively) in surface water samples. Aquatic organisms are not predicted to be at risk from water column exposures. Toxicity bioassays were not conducted with surface water in Spring Creek and they are not warranted given the low measured levels.

b) Sediments and Benthic Macroinvertebrates

Measured levels of mirex were well below the estimated sediment toxicity threshold (the quotient analysis ratio was 0.08) and therefore negligible risks from mirex are expected for benthic organisms. Marginal risks are estimated for kepone (ratio of 1.8) in sediments.

Sediment bioassays were conducted with sediment collected in the same area as for the RI chemical analysis. These sediments did not affect survival of either amphipods or midges. There were statistically significant reductions in growth of midge (30 percent) in the three highest treatment levels.

However, there is some uncertainty regarding the ecological significance of these bioassay results. It is currently unknown if growth depressions in the range observed in these bioassays would result in reduced productivity or survival, or would otherwise affect the structure and function of natural populations of benthic organisms. Growth in this species of midge does appear to be somewhat variable in the laboratory, and the ranges of midge weights in all treatments were, in fact, very similar. Therefore, it is possible that the variations in growth observed in the bioassay would not translate into an ecologically significant response, although this cannot be stated definitively.

The historical benthic macroinvertebrate data provide somewhat equivocal data for assessing benthic impact (Figure 13; *Immediate Vicinity of Route 26; 100' to 1000' Downstream of Route 26, and Upstream from Slab Cabin Run*). The species data show seasonal and annual variation, but generally there is a mix of pollution tolerant and pollution intolerant species in Spring Creek between Thornton Spring, and the Route 26 drainage ditch, and Houserville Park (upstream of Slab Cabin Run). The most recent data (1986) suggests that there are no appreciable differences between these areas and the area *Directly Upstream from the Thornton Spring Confluence.*

The broad conclusion put forward by PADER in their 1990 Water Quality Standards Review, and supported by their field surveys, states that "Spring Creek is characterized by a relatively low number of benthic macroinvertebrate taxa but shows a high relative abundance of individuals which is typical of productive, fertile limestone streams". This conclusion is further supported by PADER's conclusion regarding the brown trout population in Spring Creek (including this same area -- see Figures 11 and 12; *0-1 Miles Downstream*), that "Spring Creek supports an excellent brown trout population ...". It seems intuitive that the benthic macroinvertebrate community (i.e., the fish food) would need to be in good condition in order to support a productive fishery, one that exceeds the Pennsylvania Fish Commission's (PFC) minimum criteria of 40 kg/hectare for Class A Wild Trout Waters.

TABLE 71
QUALITATIVE COMPARISON
OF BENTHIC MACROINVERTEBRATES IN FISHING CREEK AND SPRING CREEK, PA

	STATIONS (Described below.)				
		SPRING CREEK			
TAXA	Fishing Creek at Loganton July 1985	Above Thornton Spring July 1985	Route 26 Bridge September 1986	Houserville July 1985	Houserville February 1992
	1	2	3	4	5
PLATYHELMINTHES (Flatworms)					
Turbellaria					
Planariidae		P	P	P	P
NEMATODA (Roundworms)					P
ANNELIDA (Earthworms, Leeches)					
Hirudinea	P				
Oligochaeta	C			P	P
HYDRACARINA (Water mites)					
Hydracarina					A
DECAPODA					P
AMPHIPODA (Scuds)					
Gammaridae		A	A		A
ISOPODA (Sow Bugs)					
Asellidae	P	C	P		P
PLECOPTERA (Stoneflies)					
Perlidae			P		
Allonarcys (?)	P				
Unidentified*	P				
EPHEMEROPTERA (Mayflies)					
Baetidae	C-A	C-A	C		C
Heptageniidae	P	P	C		P
Ephemerellidae	C	P	P		
Leptophlebiidae		P			P
Ephemeridae					C
ODONATA (Dragonflies, Damselflies)					
Gomphidae	P				
MEGALOPTERA (Alderflies, Dobsonflies, Fishflies)					
Corydalidae	P				
TRICHOPTERA (Caddisflies)					
Hydropsychoidea			A		A
Hydropsychidae	C	C	C		P
Rhyacophilidae	P	P	P		P
Glossosomatidae	C	P			
Hydroptilidae	C				
Brachycentridae		P			
Limnephilidae	P				

* This stonefly genus was noted as unidentifiable due to poor sample condition.

TABLE 71
QUALITATIVE COMPARISON
OF BENTHIC MACROINVERTEBRATES IN FISHING CREEK AND SPRING CREEK, PA

	STATIONS (Described below.)				
		SPRING CREEK			
	Fishing Creek at Loganton July 1985	Above Thornton Spring July 1985	Route 26 Bridge September 1986	Houserville July 1985	Houserville February 1992
TAXA	1	2	3	4	5
COLEOPTERA (Beetles)					
Elmidae	C	P	P		A
DIPTERA (Midges, Flies)					
Tipulidae	P	C	P	A	A
Ceratopogonidae	C				
Simuliidae	C	C	A	P	A
Chironimidae	C	A	C	A	A
Empididae	P		P		P
Muscidae	C				
GASTROPODA (Snails, Limpets)					
Ferrisia			P		
Physa	P				
Planorbidae	P				

RELATIVE ABUNDANCE

A = Abundant C = Common P = Present

Relative Abundance values for Spring Creek stations 2 and 4 were calculated by averaging the results from three kick samples taken at each sample point. Values were categorized according to the following scale:

A = >25
C = 10 - 25
P = 1 - 10

STATION IDENTIFICATION:

1: Fishing Creek, Loganton, PA, 200 feet downstream of Route 477. (July, 1985)

Data for Fishing Creek was obtained from Appendix F, Benthic Macroinvertebrates Qualitative, of the July, 1985 PADER study.

2: Spring Creek, at Mile Marker 16.1 upstream from Bald Eagle Creek; in Lemont, 0.5 mile upstream from PA 26 bridge and Thornton Spring. (July, 1985)

3: Spring Creek, at Route 26 bridge. (September, 1986)

4: Spring Creek, at Mile Marker 15.2 upstream from Bald Eagle Creek; in Houserville, 0.5 mile downstream from PA 26 bridge and Thornton Spring, and 0.5 mile upstream from Slab Cabin Run. (July, 1985)

5: Spring Creek, at Mile Marker 15.2 upstream from Bald Eagle Creek; in Houserville, 0.5 mile downstream from PA 26 bridge and Thornton Spring, and 0.5 mile upstream from Slab Cabin Run. (February, 1992)

Data for Spring Creek, stations 2 and 4, was obtained from a Bureau of Water Quality Management Aquatic Biological Investigation, Spring Creek, Stream File 4.20.3, Centre County, September 1984 - September 1985.
Data for Spring Creek, station 3, was obtained from the September, 1986 PADER study on Spring Creek.
Data for Spring Creek, station 5, was obtained from the February, 1992 PADER study on Spring Creek.

As an additional means to evaluate potential impacts on the benthic macroinvertebrate community in Spring Creek, a comparison has been made between the Spring Creek data and similar surveys conducted by PADER in Fishing Creek. Fishing Creek is a limestone stream within Clinton County and was suggested as a suitable comparison by the PADER Regional Aquatic Biologist. Data from four sample events conducted by PADER in Fishing Creek between 1972 and 1985 were evaluated to determine the most appropriate comparison to Spring Creek. Table 71 presents a comparison of the benthic macroinvertebrate taxa observed in Spring Creek with those present in Fishing Creek. For the purpose of this comparison, the most recent data (July 1985) from Fishing Creek were utilized together with data for Spring Creek in areas upstream of Thornton Spring and within the Houserville area, downstream of Thornton Spring. In addition, the data from September 1986 is presented to show conditions at the Route 26 bridge immediately downstream of Thornton Spring at the most comparable date available. Data from February 1992 are also presented to show current conditions and to reflect the variability in the data collected for Spring Creek.

As shown on the table, impacts are discernable in the 1985 sampling event at Houserville, however, the 1986 data (from the Route 26 bridge) and the 1992 data (from Houserville) indicate healthy aquatic conditions. Overall, the data show that most of the aquatic orders identified were found within both stream systems. The data also suggest that Plecoptera, while present in Fishing Creek, are less prevalent in Spring Creek even upstream of Thornton Spring. However, Ephemeroptera and Trichoptera have consistently been observed in both creeks, indicating generally healthy aquatic conditions. The data suggest that benthic macroinvertebrates present within Spring Creek are typically within the expected taxa when compared with Fishing Creek as a "reference stream".

It should be noted that the level of resolution for comparing benthic macroinvertebrate data between the two streams is limited by the "unknown" environmental factors that can influence the presence and abundance of taxa. There could be important habitat differences such as substrate and stream morphology, and flow velocity, along with differences in water quality. Differences in the data could also be a function of sample collection

differences, as well as seasonal factors which, as shown in Figure 13, are considerable in Spring Creek.

c) Fish

There are no available data to correlate fish tissue concentrations of mirex or kepone with toxicity to the fish themselves. The quotient method analysis does not predict risk to fish from the water or the sediment, but the best information for the assessment of risk comes from the historical data on the trout populations in Spring Creek.[1] As noted in the previous section, the historical data and the PFC's Class A Wild Trout Water criteria both support the conclusion that the mirex and kepone (plus any other anthropogenic chemicals in Spring Creek) are not having a significant adverse impact on the fish. This conclusion is based on the absolute numbers and biomass of the brown trout fishery along the 25-mile stretch of Spring Creek as well as by comparison of the fishery in Spring Creek to other brown trout fisheries around the country.

There are numerous influential/controlling factors that are important when attempting to compare fisheries from different locations. These include habitat conditions, fishing restrictions (lures, limits), stocking and competition of wild and stocked fish. For the purpose of assessing the overall condition of the fishery in response to a perturbation such as chemical contamination, a general comparison such as fish density (numbers and biomass) may be most useful. Carline (1991) of the Pennsylvania Fish Commission discusses the density of brown trout in six streams including Spring Creek. Table 72 lists the streams, some of the controlling factors, and the trout density (numbers per hectare). Again, there are reasons why the fish density will vary between different sites, but each of these streams is considered a very good trout fishery; Spring Creek ranks fourth among these streams with respect to trout density.

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[1] While there are no similar data for other fish species, neither are there any data to suggest that other fish species would be more sensitive than trout. In fact, because trout are carnivores, they have a relatively high potential to be exposed to chemicals that accumulate through the aquatic foodchain.

TABLE 72
Population Density and Management Plan for Adult (Brown Trout in Various U.S. Streams)

Stream	Trout Species	Stocked	Bag Limit	Lure Restriction	Trout Density (Numbers/ Hectare)	Reference
Spring Creek, PA	Brown	No	0	Flies only on certain sections	725[1]	Carline 1991
Fishing Creek, PA	Brown and Brook	Yes	2	Artificial lures only	270	PADER 1986
Castle Rock Creek, WI	Brown	Yes	0	Artificial lures only	450	Hunt 1987 as cited in Carline 1991
Silver Creek, Idaho	Brown	No	0	Flies only	44	Rhiele et al. 1989 as cited in Carline 1991
Timber Coulee Creek, WI	Brown and Brook	Yes	1	Artificial lures only	1,030	Hunt 1987 as cited in Carline 1991
Race Branch, WI	Brown and Rainbow	Yes	1	Artificial lures only	1,757	Hunt 1981 as cited in Carline 1991
Hot Creek, CA	Brown and Brook	No	2	Flies only	1,928	Dienstadt 1977 as cited in Carline 1991

NOTES:

[1] This value represents the overall mean population density from the most recent sampling (1988)

In addition, PADER has provided fish population information on Fishing Creek, another Pennsylvania limestone stream with good water quality and a thriving trout population. There are very limited historical data on trout densities in Fishing Creek. A 1985 study does however provide density data for one sampling station in Fishing Creek (PADER 1986). The combined number of adult brown and brook trout at this station was 270 per hectare, and the biomass estimate was over 47 kg per hectare. While these densities are somewhat lower than the most recent 1988 data for Spring Creek (i.e., 725 adult brown trout per hectare, and 143 kg per hectare), there are differences in the two creeks that would contribute to such differences. Spring Creek is not stocked, while Fishing Creek is stocked with both brown and brook trout. Also, Spring Creek is under a no-harvest advisory, while at Fishing Creek, two trout may be legally creeled per day. Despite their differences, both creeks exceed the minimum criteria of 40 kg/hectare set by the Pennsylvania Fish Commission for Class A Wild Trout Waters and the available sampling data are indicative of thriving trout populations.

Finally, the level of mirex detected in upper trophic level fish collected during the RI in the RMU5 (Houserville Park) area exceed the FDA action level for human consumption. The FDA action level is 0.1 ppm mirex and the brown trout fillet measured 0.17 ppm. The trout sample had no detectable level of kepone. The quantitation limit was 0.19 ppm and the kepone action level is 0.3 ppm. The slimy sculpin tissue levels were approximately 0.3 ppm mirex and 0.5 ppm kepone, but these whole body analyses are not relevant for human consumption. The historical fish tissue data for the Houserville area shows generally declining levels of mirex with time (Figure 10).

d) Piscivorous Wildlife

The quotient analysis predicts negligible risks to sensitive piscivorous wildlife (i.e., mink, heron and kingfisher) that might be exposed to mirex or kepone via consumption of fish, aquatic invertebrates, and sediment. The estimates are based on RME levels of mirex and kepone. The risk quotients range from .02 to .37.

6. RMU6 (Spring Creek downstream in the vicinity of the Benner Hatchery)

Aquatic organisms are not predicted to be at risk in the water column. Both mirex and kepone were below detection limits in surface water samples.

The quotient analysis predicts that benthic organisms are not at risk from mirex in sediments; the quotient, based on RME mirex levels, is .04. The quotient (4.6) for kepone in sediments infers a risk to sensitive species. However, the discussions of benthic macroinvertebrate and brown trout survey data for RMU5 also apply to RMU6 (also see Figures 11-13 for data from the Benner Hatchery area). These populations appear to be in good condition overall, supporting a conclusion that mirex and kepone in the sediment (or in fish tissue) are not having a significant ecological impact on the benthos or fish.

As with the two upstream sampling locations of Spring Creek, RMU6 has negligible potential for risk to piscivorous wildlife that might consume all of their diet at the RME levels of mirex or kepone in fish, invertebrates, etc. This means that whether the resident species stay in a relatively small area of the Creek or they move up and down stream, there are negligible risks predicted from this ecological assessment.

Finally, the level of mirex detected in upper trophic level fish collected during the RI in the RMU6 (Benner Spring) area is right at the FDA action level for human consumption. The action level is 0.1 ppm mirex and the brown trout fillet measured 0.11 ppm. The trout sample had no detectable level of kepone. The quantitation limit was 0.19 ppm and the kepone action level is 0.3 ppm. The slimy sculpin tissue levels were approximately 0.2 ppm mirex and 0.5 ppm kepone, but these whole body analyses are not relevant for human consumption. The historical fish tissue data for the Benner Spring area shows the levels of mirex and kepone generally declining with time (Figure 10).

7. Conclusion

In summary, this ecological risk assessment, takes into account all the available information and analyses including: 1) the results of the quotient method analysis; 2) the surface water and sediment bioassay data; 3) the historical monitoring of benthic macroinvertebrates, fish, and fish tisssue levels of mirex and kepone; 4) the comparison of the Spring Creek aquatic community to other comparable streams; and, 5) the observations of biota made during the RI sampling. These data support the following conclusions:

- There are negligible risks to both the biota that inhabit the 15-acre field adjacent to the Ruetgers-Nease facility, and the biota that inhabit the Spring Creek area (both aquatic and terrestrial species).

  - While the RI sampling, bioassays, and quotient ratio analyses are important components of a thorough assessment of risk, the actual conditions in the field provide the best integration of any significant exposures and impacts that might be occuring. The twenty plus years of survey data for Spring Creek have documented both healthy and impacted aquatic habitats at various times in the same areas. The impacts have been sporadic and are clearly not solely attributable to a single source such as the Nease Chemical Site or the Ruetgers-Nease facility. One of the best available measures of the current (and historic) condition of Spring Creek is that it supports a very productive trout fishery, which indirectly indicates an overall healthy benthic macroinvertebrate community (i.e., fish food).

- There are measureable levels of mirex and kepone in the fish in Spring Creek. The current levels are at or just above at the FDA action levels for human consumption. The historical data show a trend of decreasing levels of mirex and kepone from fish collected in the same areas of Spring Creek sampled during the RI. The levels in fish generally decrease with increasing distance downstream.

- While there are detectable levels of mirex and kepone in the trout, there does not appear to be an adverse impact on the trout population. Historical and current data support a trout population that exceeds the State's minimum criteria for Class A Wild Trout Waters of 40 kg/hectare.

- Both the quotient approach and the bioassays predict some risks to sensitive aquatic biota that would inhabit the surface water of the drainage ditch and the sediments of the on-site drainage ditch as well as Thornton Spring. However, the magnitude of the risks may be overestimated by these two assessment tools.

- Because of the intermittent water flow, the drainage ditch is not a true aquatic environment and sensitive species may not inhabit the drainage ditch (i.e., fish are not inhabitants). Thornton Spring, although limited in size and habitat diversity, appears to provide a habitat capable of supporting a "more typical" aquatic biota, exclusive of fish. Midge, earthworms, and crayfish were observed in Thornton Spring during the RI sampling, suggesting the magnitude of the risks are limited. The toxicity observed in bioassays conducted using the drainage ditch and Thornton Spring surface waters would not be expected to extend to the surface waters of Spring Creek. Sediments collected in Spring Creek just below the confluence of Thornton Spring were not toxic.

- There are negligible aquatic foodchain risks associated with Thornton Spring and/or the drainage ditch because neither area would support piscivorous species. Negligible risks were predicted for terrestrial species that might prey on soil/sediment invertebrates from these areas.

- Finally, no animal species of special concern are believed to reside within five miles of State College, an area which includes the entire Site.

G. Uncertainties in the Analysis

As in any risk assessment, this assessment contains elements of uncertainty that limit the degree to which absolute conclusions can be made regarding ecological risks. Many of these uncertainties stem from an incomplete knowledge of the degree of exposure to ecological receptors and the toxicological and ecological significance of the exposures. Considerations incorporated into this ecological assessment to reduce uncertainties include analysis of the fish tissue residue data and performance of surface water and sediment bioassays. In addition, over 20 years of historical data from a number of sources have been used to help define the distribution of site-related chemicals within Spring Creek. Despite these considerable data, and applying the available guidance for conducting an ecological risk assessment for an RI, some uncertainties exist. The most significant sources of uncertainty associated with this assessment are discussed below.

1. Distribution of Mirex and Kepone in Spring Creek

Conclusions in this report regarding the distribution of mirex and kepone in Spring Creek are based primarily on analytical data for surface water and sediment collected as part of the Remedial Investigation, as well as historical studies conducted in Spring Creek by PADER, PFC, and the USEPA. Collectively, these studies provide chemical data for 20 locations throughout Spring Creek, beginning in areas upstream of the Site and continuing to a point downstream of the confluence of Spring Creek with Bald Eagle Creek, as well as sample locations in Blanchard Lake (into which Bald Eagle Creek flows). These data are supplemented by analytical data on mirex and kepone concentrations in fish tissue, which provide evidence of the bioavailability of these chemicals.

Overall, these data indicate that mirex and kepone concentrations are limited spatially to stream areas closest to the Site (the concentrations of both chemicals in sediment and fish decrease with increasing downstream distance from the site). Further, the data suggest that bioavailability is decreasing with time (the fish tissue data generally show an overall downward trend). These general conclusions are consistent with the data presently available, but uncertainties still remain with respect to the vertical and horizontal distribution of mirex and kepone in the Spring Creek system. From a risk assessment standpoint, however, the concentrations in the Spring Creek system would have to be substantially higher than those measured for the conclusions of the risk assessment to be changed significantly. It is considered unlikely that concentrations of mirex and kepone in the Spring Creek system would exceed the levels previously detected given that: 1) over 16 years have passed since mirex or kepone were manufactured at the Site and the principal chemical releases occurred; 2) several interim remedial measures have been conducted to limit further releases; and 3) all data support the position that chemical concentrations are greatest near the Site.

2. Complete Ecological Characterization versus Indicator Species

The approach used to characterize ecological risks in this assessment was based on the indicator species concept. Under this approach, risks are estimated for certain components of the ecological community under the assumption that effects in other portions of the community will be equal to or less than those predicted. The indicator species approach is generally used in ecological risk assessment since it is not feasible to assess risks or measure impacts in all components of the system. The

key to the indicator species approach is to address all the major exposure pathways, and to select components of the system that are the most susceptible or sensitive to impacts and also that are sufficiently important resources from a risk management standpoint to warrant action if risks are predicted.

The indicator groups chosen for this assessment are believed to meet these important criteria. For example, brown trout was used as an important indicator species for the aquatic system. This species is likely susceptible to impacts from mirex or kepone given that it is at the top of the food chain (and therefore could experience the greatest exposures) and that it is a member of a family (*Salmonidae*) that is often very sensitive to chemical toxicants. In addition, because of its position at the top of the aquatic food chain, trout are likely a good integrator of stresses to all components of the system. For example, if the benthic community is severely affected, the trout population would likely be affected because of impacts on its food resource. Brown trout also are believed to be a good indicator species for the Site because they are an important ecological resource that is highly valued from a fishery standpoint. The other receptors selected for this assessment (i.e., piscivorous and non-piscivorous wildlife, and the sensitive bioassay species) are also believed to meet the criteria for good indicator species.

Nevertheless, it is possible that other components of the system could be more sensitive or susceptible to impacts than those evaluated, and therefore, that these other resources may be at risk. The approach used here is the most appropriate in that both sensitive and integrator species are addressed to evaluate the risks to the system as whole. The assumption is made that since these species do not appear to be impacted, it is because either 1) none of the components of the system are being impacted, or 2) any impacts on other components that are occurring are not sufficient to affect the system as a whole. These assumptions have attendant uncertainties that cannot be reduced at this time.

3. Implications of Sediment Bioassay Results to the Benthic Community and Fish Residue Data to Fish Populations

There is uncertainty in extrapolation of the sediment bioassay results to the benthic community. The only adverse effect observed in the sediment bioassay was a statistically significant reduction in growth of the midge in sediment collected at the Houserville Park location; however, the most recent (February 1992) PADER monitoring data indicates the benthic macroinvertebrate community is healthy in the Houserville Park area. There is also uncertainty in extrapolation of the fish body

burden data to the health of the fish. There Spring Creek trout and sculpin populations are healthy, although both species have measurable levels of mirex and kepone in their tissues. The state of knowledge in this field does not allow for a more direct extrapolation of the bioassay and tissue residue results to predict risks to the invertebrates and fish. Generally the bioassays are used to screen for potential adverse effects and the fish tissue data are used to estimate exposures up the food chain. If adverse impacts are predicted, then biomonitoring of the condition of populations can be used as a more direct measure of impacts. The historical benthic macroinvertebrate and brown trout population data provide that more direct indication of the ecological significance of the mirex and kepone present in the Spring Creek system. In the absence of definitive predictions or observations of significant adverse ecological impacts, monitoring for trends/changes may provide the best direct assessment for the Site.

4. Methodological Uncertainties with the Quotient Evaluation

A combination of site-specific chemical concentration data and literature on bioaccumulation and toxicity was used in the risk analysis. The quotient method was used to provide the best available level of quantitation to the analysis. The more obvious uncertainties that are inherent in the quotient method approach for estimating risk are summarized below.

a) Toxicity Estimates

The chronic toxicity thresholds for pisciverous and other predatory animals are based on the results of laboratory tests with representative species which are not necessarily those associated with the spray field, the drainage ditch, Thornton Spring, or Spring Creek. By design, the threshold calculations are intended to protect the most sensitive species. Conservative application and uncertainty factors are built into the calculations from the outset. The considerable toxicity data for birds and mammals for both mirex and kepone improved the extrapolation.

b) Exposure Estimates

It was assumed that the selected receptor species are exposed continuously and over long periods of time to the RME levels of a particular chemical. Thus both the levels and duration of exposure are conservatively

derived. The actual exposure is likely to be less rather than more than what was assumed in the exposure model.

c) Quotient Values and Risk Magnitudes

The quotient values (i.e., the magnitude of the ratio between the chemical exposure estimate and the chronic toxicity threshold) are intended to correlate with the potential for, and magnitude of, risk. It is assumed that the level of risk and the magnitude of adverse effects increase as the exposure exceeds the toxicity threshold. There are, however, no ecological standards comparable to the human health standards. The general guidance indicates that a quotient greater than one (1) infers a potential risk and a quotient of less than one infers no risk (Rodier and Maurielio 1993). There is an available guideline for interpreting the quotients for piscivorous birds and mammals (which is relevant to this Site). Newell et al. (1987) suggests that a quotient exceeding ten (10) is likely to relate to a high potential for risk.

When there is a predicted exceedence of exposure compared to a toxicity threshold, better site-specific exposure and/or toxicity information may be needed to reduce the uncertainty associated with the assumptions that went into the analysis. The site-specific bioassays with aquatic invertebrates and fish, and the population monitoring data for the benthic macroinvertebrates and fish, available in this case, are examples of the types of site-specific data that can be used to reduce the uncertainties inherent in the quotient approach. In the absence of generally accepted standards, professional judgement must be used to determine whether enough information exists to weigh the uncertainties and to make risk management decisions.

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